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CORNELL-DUBILIER ELECTRONICS SUPERFUND SITE
SOUTH PLAINFIELD, NEW JERSEY

FINAL
TECHNICAL IMPRACTICABILITY EVALUATION
OPERABLE UNIT 3: GROUNDWATER

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List of Acronyms and Abbreviations

°C	Degrees Celsius
°F	Degrees Fahrenheit
ARAR	Applicable or Relevant and Appropriate Requirements
ATSDR	Agency for Toxic Substances and Disease Registry
bgs	below ground surface
BHHRA	Baseline Human Health Risk Assessment
CDA	Capacitor Disposal Area
CDE	Cornell-Dubilier Electronics, Inc.
CEA	Classification Exception Area (for groundwater)
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
cis-DCE	cis-1,2-Dichloroethene
CM	Corrective Measures
COC	Chemical of Concern
COPC	Chemical of Potential Concern
CTE	Central Tendency Exposure
CVOC	Chlorinated Volatile Organic Compound
CY	Cubic Yard
DFN	Discrete Fracture Network
DHE	Dehalococcoides Ethenogenes
DNAPL	Dense Non-Aqueous Phase Liquid
DO	Dissolved Oxygen
DUS	Dynamic Underground Stripping
EPM	Equivalent Porous Medium
ERH	Electrical Resistance Heating
ERT	Environmental Response Team (USEPA)
EVO	Emulsified Vegetable Oil

FS	Feasibility Study
gpm	Gallons per minute
GRA	General Response Action
HI	Hazard Index
HPO	Hydrous Pyrolysis Oxidation
HQ	Hazard Quotient
ISCO	In Situ Chemical Oxidation
ISTD	In Situ Thermal Desorption
LBG	The Louis Berger Group, Inc.
LTTD	Low-Temperature Thermal Desorption
msl	Mean sea level
NAWC	Naval Air Warfare Center (former)
NCP	National Contingency Plan
NJAC	New Jersey Administrative Code
NJDEP	New Jersey Department of Environmental Protection
NJPDES	New Jersey Pollutant Discharge Elimination System
NPL	National Priorities List
O&M	Operations and Maintenance
OMB	Office of Management and Budget
ORP	Oxidation Reduction Potential
OSWER	Office of Solid Waste and Emergency Response
OU	Operable unit
PAH	Polycyclic aromatic hydrocarbon
PbB	Blood Lead Level
PCB	Polychlorinated biphenyl
pg/L	Picogram per Liter
POTW	Publicly Owned Treatment Works
PRB	Permeable Reactive Barrier
PRG	Preliminary Remediation Goal

RAGS	Risk Assessment Guidance for Superfund
RCRA	Resource Conservation and Recovery Act
RI	Remedial Investigation
RME	Reasonable Maximum Exposure
ROD	Record of Decision
SARA	Superfund Amendments and Reauthorization Act of 1986
SEAR	Surfactant Enhanced Aquifer Remediation
SEE	Steam-Enhanced Extraction
SVE	Soil Vapor Extraction
SVOC	Semi-volatile organic compound
TBC	To Be Considered
TCDD	Tetrachlorodibenzo-*p*-dioxin
TCE	Trichloroethylene; trichloroethene
TCH	Thermal Conduction Heating
TDA	Temporary Discharge Approval
TEQ	Toxic equivalence
TI	Technical Impracticability
TIER	Technical Impracticability Evaluation Report
TOC	Total Organic Carbon
TSDF	Treatment, Storage, and Disposal Facility
USACE	United States Army Corps of Engineers
USEPA	United States Environmental Protection Agency
VOC	Volatile organic compound
WVA	Watervliet Arsenal

Introduction

This Technical Impracticability Evaluation Report (TIER) presents the justification for the waiver of Applicable or Relevant and Appropriate Requirements (ARARs) for Operable Unit 3 (OU 3) of the Cornell-Dubilier Electronics (CDE) Superfund Site (Site) [EPA ID: NJD981557879] located in South Plainfield, New Jersey. OU3 addresses the contaminated groundwater portion of the Site. This TIER has been prepared in accordance with the United States Environmental Protection Agency (USEPA) Office of Solid Waste and Emergency Response (OSWER) Directive 9234.2-25, *Guidance for Evaluating the Technical Impracticability of Groundwater Restoration* (TI Guidance) (USEPA 1993), and on behalf of the U. S. Army Corps of Engineers (USACE), Kansas City District and the USEPA Region II.

In accordance with the provisions in Section 4.2 of the TI Guidance, this TIER is being submitted after completion of the Remedial Investigation (RI Report) as the data collected during the RI are sufficient to identify the critical limitations to groundwater restoration. Accordingly, this "front-end" TIER demonstrates the impact of these critical limitations on contaminant distribution, restoration potential, and the effectiveness of currently available remedial technologies.

Site Location and Background

Cornell-Dubilier Electronics, Inc. operated from 1936 to 1962, manufacturing electronic parts and components, including capacitors. The company released material contaminated with polychlorinated biphenyls (PCBs) and other hazardous substances, including chlorinated solvents, directly onto the soil during its operations. USEPA detected PCBs and chlorinated solvents in the groundwater and soil at the former CDE facility and at nearby residential, commercial and municipal properties. USEPA also detected PCBs in the surface water and sediments of Bound Brook, which is adjacent to the former CDE facility's southeast corner. The Site has been divided into four Operable Units (OUs) by the USEPA. Operable Unit 1 (OU1) addresses residential, commercial, and municipal properties in the vicinity of the former CDE manufacturing facility (the former CDE facility) at 333 Hamilton Boulevard. The USEPA signed a Record of Decision (ROD) for OU1 in 2003. Operable Unit 2 (OU2) addresses contaminated soil and buildings at the former CDE facility. The USEPA signed a ROD for OU2 in 2004. OU3 addresses contaminated groundwater and Operable Unit 4 (OU4) addresses Bound Brook.

Justification for Technical Impracticability

There are significant Site-specific factors that limit the ability of available remedial technologies to achieve groundwater ARARs at the Site. These factors include the long history of releases, the presence of dense non-aqueous phase liquids (DNAPL) and chlorinated volatile organic compounds (CVOCs) in the bedrock groundwater and rock matrix porewater, and the complex geology of the Site. In addition, the likely presence of outside sources of CVOC contamination in the area of the Site also contributes to the technical impracticability (TI) determination as their presence precludes restoration of the groundwater to ARARs. This determination is supported by the following:

1. The highly conductive fracture network at the Site allows for the vertical and horizontal advection of groundwater and aqueous-phase contaminant mass. Because the fracture network is so pervasive, it provides a relatively large surface area for the CVOCs to sorb onto and then diffuse into the rock matrix. The pore volume of the rock matrix is nearly two orders of magnitude larger than the fracture network, allowing it to hold the majority of the contaminant mass. Once the aqueous-phase contaminant mass diffused into the rock, it was left nearly immobile because of the low hydraulic conductivity of the rock matrix. Back diffusion out of the matrix (pore water) is occurring in areas where the concentration gradient between the rock matrix and the aqueous phase in fractures supports the process, which will contribute to ongoing groundwater contamination over a very long period of time. As a result, the contaminated aquifer cannot be restored such that it meets ARARs, allowing use as a potable water supply without treatment at the wellhead, in a reasonable timeframe or at a reasonable cost.

2. Samples of the bedrock matrix and groundwater show that CVOCs have adsorbed into the bedrock matrix over a very large (~150 acres) area; the highest observed concentrations of CVOCs in the bedrock matrix are located at MW-14S/D on the former CDE facility.

3. Contaminant fate and transport modeling results indicate that treatment of bedrock limited to the area beneath the overburden source area (i.e., at MW-14S/D) would have negligible impact on the remainder of the downgradient plume and would not result in the achievement of ARARs since the bedrock matrix itself is the source of the ongoing exceedence of ARARs. Therefore, to be potentially capable of meeting ARARs, a remedial technology would have to be applied over the *entire* OU3 area where bedrock matrix contamination contributes to ongoing exceedences of ARARs. This would encompass an area of more than 150 acres and depths of more than 250 feet below ground surface. The implementation of any of in-situ remediation technology over such an area is not practicable.

4. To be successful, a remedial technology would have to be capable of treating contamination in the rock matrix and the bedrock fractures. To do this, the technology must be capable of contacting impacted areas and maintaining that contact over a long enough time period to promote treatment in the rock matrix. Based on the review of currently available remedial technologies, there are no

technologies capable of achieving these metrics in fractured bedrock in full-scale implementation.

TI Zone

The USEPA TI Guidance (USEPA 1993) states that at sites where restoration of groundwater to its most beneficial use is technically impracticable, the area over which the decision applies (referred to as the TI Zone) generally will include all portions of the contaminated groundwater that do not meet ARARs. ARARs are waived inside the TI Zone and other measures, such as pathway elimination and/or administrative controls, are used to prevent exposure to human health and the environment. Outside of the TI Zone, ARARs will still apply. In accordance with the TI Guidance, a TI Zone has been developed that meets these criteria (see Figure ES-1).

1. Introduction

1.1. Purpose

This TIER presents the justification for the waiver of ARARs for OU3 of the CDE Superfund Site [EPA ID: NJD981557879] located in South Plainfield, New Jersey. OU3 addresses the contaminated groundwater portion of the Site. This TIER has been prepared in accordance with the USEPA OSWER Directive 9234.2-25, *Guidance for Evaluating the Technical Impracticability of Groundwater Restoration* (TI Guidance) (USEPA 1993), and on behalf of the USACE, Kansas City District and the USEPA Region II.

An ARAR waiver is sought when site-specific conditions render it technically impracticable, from an engineering perspective, to achieve those ARARs within a reasonable timeframe and at a reasonable cost. Site-specific factors such as the long history of releases, the presence of DNAPL and the widespread presence of CVOCs in the bedrock groundwater and rock matrix porewater, and the complex geology of the Site contribute to the impracticability of groundwater restoration. In addition, the likely presence of outside sources of CVOC contamination in the area of the Site can also contribute to a TI determination as their presence precludes restoration of the groundwater to ARARs. Therefore, consistent with EPA guidance, the appropriateness of an ARAR waiver is being evaluated for the Site in connection with the restoration of groundwater.

In accordance with the provisions in Section 4.2 of the TI Guidance, this TIER is being submitted after completion of the Remedial Investigation (RI Report) as the data collected during the RI are sufficient to identify the critical limitations to groundwater restoration. Accordingly, this "front-end" TIER demonstrates the impact of these critical limitations on contaminant distribution, restoration potential, and the effectiveness of currently available remedial technologies.

1.2. Background and Setting

Cornell-Dubilier Electronics, Inc. operated at 333 Hamilton Boulevard, South Plainfield, New Jersey from 1936 to 1962, manufacturing electronic parts and components, including capacitors. The company released material contaminated with PCBs and trichloroethene (TCE) directly onto the soils during its operations. USEPA has detected PCBs and CVOCs in the groundwater and soil at the former CDE facility and at nearby residential, commercial and municipal properties. USEPA also has detected PCBs and CVOCs in the surface water and sediments of Bound Brook, which is adjacent to the former CDE facility's southeast corner. The Site has been divided into four OUs by the USEPA. OU1 addresses residential, commercial, and municipal properties in the vicinity of the former CDE facility at 333 Hamilton Boulevard. The USEPA signed a ROD for OU1 in 2003. OU2 addresses contaminated soils and buildings at the former CDE facility. The USEPA signed a ROD for OU2 in 2004. OU3 addresses contaminated groundwater and OU4 addresses Bound Brook.

As such, the following terminology will be used throughout this report:

The "**Site**" refers to all four OUs which comprise the CDE Superfund Site, and the extent of each OU investigation;

The term **"off-Site"** refers to any area that is beyond the limits of the former CDE facility (OU2); off-Site areas may still be within the Site.

The "**former CDE facility**" refers to the physical extent of the industrial park operated at 333 Hamilton Boulevard, which is OU2; and

"**OU3**" refers to the geographic extent of the groundwater contamination and associated investigation.

The former CDE facility is located at 333 Hamilton Boulevard in South Plainfield, Middlesex County, New Jersey (Figure 1-1) and covers approximately 26 acres. Most recently, the property was known as the Hamilton Industrial Park. It contained numerous buildings. These buildings were demolished in 2008 following relocation of the industrial park tenants.

The Spicer Manufacturing Company operated a manufacturing plant on the property from 1912 to 1929. They manufactured universal joints and drive shafts, clutches, drop forgings, sheet metal stampings, screw products, and coil springs for the automobile industry. The plant included a machine shop, box shop, lumber shop, scrap shop, heat treating building, transformer platform, forge shop, shear shed, boiler room, acid pickle building, and die sinking shop. A chemical laboratory for the analysis of steel was added in 1917. Most of the major structures were erected by 1918. When the Spicer Manufacturing Company ceased operations at the facility, the property consisted of approximately 210,000 square feet of buildings (FWENC, 2002). Even though TCE was commercially available during the latter half of Spicer Manufacturing Company's period of operation on the property, there is no documentation that TCE was used in the manufacturing process during their period of operation on the property.

After the departure of the Spicer Manufacturing Company, CDE manufactured electronic components, including capacitors, from 1936 to 1962. It has been reported that the company also tested transformer oils for an unknown period of time. PCBs and chlorinated organic degreasing solvents were used in the manufacturing process, and the company disposed of PCB-contaminated materials and other hazardous substances at the facility. It has been reported that the rear of the property was saturated with transformer oils and capacitors were also buried behind the facility during the same time period (FWENC, 2002).

Since CDE's departure from the facility in 1962, it had been operated as a rental property consisting of commercial and light industrial tenants. Numerous tenants have occupied the complex. In 2007, USEPA began implementing the OU2 ROD with the relocation of the tenants at the industrial park and demolition of the 18 buildings. Relocation of the tenants was completed in mid-2007, demolition of buildings was completed in May 2008, and OU2 soil remedial activities are ongoing. A plan view of the former CDE facility, showing the location of former buildings, is included as Figure 1-2. Previous investigations have included groundwater sampling, subsurface soil sampling, sediment sampling, building surface sampling, soil gas sampling, indoor air sampling, and hydrogeological studies.

The developed portion of the facility (the northwestern portion) comprised approximately 45 percent of the total land area and contained temporary asphalt capping following building demolition, a system of catch basins to channel storm water flow, and paved roadways. Several of the catch basins drained into a storm water collection system whose outfalls discharge at various locations along Bound Brook. The other 55 percent of the property was predominantly vegetated before the OU2 remedial activities began. The central part of the undeveloped portion was primarily an open field, with some wooded areas to the northeast and south, and a deteriorated, partially paved area in the middle of the undeveloped portion of the facility. The northeast and southeast boundaries consist primarily of wetland areas adjacent to Bound Brook, which flows from the eastern corner across the northeastern border of the undeveloped portion of the facility (FWENC, 2002). Once OU2 remedial activities are completed, the entire former CDE facility will be covered by an asphalt cap with a storm water collection system.

1.3. Site Location

The Site is located in the Borough of South Plainfield, northern Middlesex County, in the central portion of New Jersey. According to the 2006 Census estimate, South Plainfield has a population of approximately 22,795 people with a total land area of approximately 8.4 square miles (City-Data.com).

The Site includes the fenced, 26-acre former CDE facility that is bounded on the northeast by Bound Brook and the former Lehigh Valley Railroad, Perth Amboy Branch (presently Conrail); on the southeast by Bound Brook and a property used by the South Plainfield Department of Public Works; on the southwest, across Spicer Avenue, by single family residential properties; and to the northwest, across Hamilton Boulevard, by mixed residential and commercial properties. The area surrounding the former CDE facility represents an urban environment with principally commercial and light industrial use to the northeast and east, principally residential development to the south and directly north, and mixed residential and commercial properties to the west.

2. Physical Characteristics of the Site

The following is a general description of the physical characteristics of the Site.

2.1. Surface Features and Topography

Prior to the OU2 remedial activities, the northwestern portion of the former CDE facility (comprising approximately 45% of the total facility acreage) contained 18 buildings that have since been demolished. The land in this northwestern portion was gently sloping, with pre-building demolition elevations ranging from 70 to 82 feet above mean sea level (msl).

The remaining 55% of the land area was undeveloped and predominantly vegetated. The central part of the undeveloped portion was primarily a flat, open field, with some wooded areas to the south. A paved area in the middle of the former CDE facility is where capacitor-related debris had been observed. This area was relatively level, with pre-OU2 remediation elevations ranging from approximately 71 to 76 feet above msl. The property drops steeply to the northeast and southeast, and the eastern portion of the property consists primarily of wetlands bordering Bound Brook. Elevations in this area ranged from approximately 71 feet above msl at the top of the bank to approximately 60 feet above msl along Bound Brook (FWENC, 2001). Ongoing OU2 soil remediation activities at the former CDE facility have altered the surface topography. At the conclusion of OU2 remedial activities, the former CDE facility will be covered by an asphalt cap, gently sloping from the southwest to the northeast; storm water will be collected by a series of catch basins and a detention basin, and will ultimately discharge to Bound Brook.

2.2. Geology

The Site lies within the Piedmont Physiographic Province of New Jersey (Fenneman, 1938). The following sections contain a brief description of the surficial and bedrock geology of the Site. More extensive information is presented in the OU3 RI report (LBG and Malcolm Pirnie, 2012).

2.2.1. Surficial Geology

Quaternary and pre-Quaternary glacial and glacial-fluvial deposits overlie bedrock across much of the northern portion of New Jersey. Based on regional surficial geologic mapping for the area, unconsolidated deposits in the vicinity of the Site include sandy, silty clay to clayey, silty sand containing some shale, mudstone, and sandstone fragments. Surficial deposits underlying the Site are generally identified as regolith derived from weathering of shale, mudstone, and sandstone. The unconsolidated deposits are up to 30 feet thick regionally, but are generally less than 10 feet thick in the vicinity of the Site (FWENC, 2002).

2.2.2. Bedrock Geology

The Site is located within the Newark Basin, which is a tectonic rift basin that covers roughly 7,500 square kilometers extending from southern New York through New Jersey and into southeastern Pennsylvania (Figure 2-1). The basin is filled with Triassic-Jurassic sedimentary and igneous rocks that are tilted, faulted, and locally folded.

The Passaic Formation (historically known as the Brunswick Formation) occupies an upper unit of the Newark Supergroup rocks in the Triassic-Jurassic Newark Basin and is the thickest and most aerially extensive unit in the Newark Basin (Herman, 2001). The Passaic Formation in the northern half of the State has been folded, faulted, and fractured during multiple tectonic events spanning hundreds of millions of years. This has contributed to the highly fractured nature of the bedrock in this area. This formation consists of mostly red cyclical lacustrine clastics including mudstone, siltstone, and shale, with minor fluvial sandstone (Michalski and Britton, 1997). The reddish color originates from reworked hematite, which comprises 5-10% of the unit. The Site is located south of the contact between the Passaic Formation mudstone unit and a thinly bedded siltstone/shale unit (Herman, 2001).

2.2.3. OU3 Geology

Unconsolidated deposits at the former CDE facility range in thickness from 0.5 to 15 feet and generally thicken to the east towards Bound Brook. Natural unconsolidated materials, consisting primarily of red-brown silt and sand with silt and clay layers, are generally intermixed with urban fill materials (including cinders, ash, brick, glass fragments, metal, and other detritus) throughout the former CDE facility and vicinity. A thin (surface to 15 feet bgs) layer of weathered bedrock overlies competent bedrock, consistent with the weathered bedrock identified by regional surficial geologic mapping. This material primarily consists of heavily weathered siltstone and shale material with a heterogeneous texture ranging from silt to fine sand, with some zones of angular, silty gravel and silty clay.

The top of competent bedrock underlying the former CDE facility ranges from 4 to 15 feet bgs, except in the northwestern portion of the former CDE facility where bedrock was present immediately beneath the former building foundations. Based on boring log data for wells installed during the OU3 RI, bedrock at the Site consists primarily of red-brown to dark brown mudstone, siltstone, and shale, consistent with the upper Passaic Formation. Boring logs from wells to the north of the former CDE facility are generally indicative of Passaic Formation mudstone facies, while cores from the facility and areas southwest and east of the facility show siltstone and shale. The bedrock units range from massive rock with few features to highly laminated beds. The bedrock units are consistently fine-grained in texture, with numerous calcified veins and vugs throughout.

Bedrock boring logs and borehole acoustical televiewer data (presented in the RI report) indicate that numerous fracture zones are present in bedrock from the surface to approximately 600 feet bgs, the maximum drilled depth. The shallow bedrock units are heavily fractured and weathered, with significant shallow fracture in-filling with weathered material ranging in texture from silt/clay to sand. Shallow fractures are generally more open in the shallow bedrock and become less open with depth. The bedrock contains heavily fractured zones that occur along the bedding planes (parallel to sub-parallel). Weathered fracture zones within the bedrock ranged from near horizontal to near vertical. Pole to plan projections of the fracture data interpreted from acoustical televiewer data (presented in the RI report) show that the majority of these features are relatively low angle, ranging from 10 to 30 degrees from horizontal, consistent with the regional character of the Passaic Formation.

Based on the borehole geophysical data collected during the OU3 RI, the bedding planes of the bedrock units (less open features) in the vicinity of the former CDE facility generally strike 63 degrees East of North (N65E), and generally dip toward the northwest between 5 and 15 degrees. The predominant down-dip direction of fractures (more open features) is toward the northwest, parallel to sub-parallel to the dip of most bedding planes. A large fracture zone was encountered in MW-14 (67 feet bgs), MW-15 (76 feet bgs), MW-17 (180 to 210 feet bgs), and downgradient from the former CDE facility at MW-20 (302 feet bgs). However, no significant fracture zone was observed in MW-16, which lies between MW-14 (near the center of the former CDE facility) and MW-20 (downgradient). The orientation of the fracture zone was calculated (3-point solution) and is nearly parallel to regional bedding. This intensively fractured seam is characterized by significantly larger than average fracture apertures.

2.3. Hydrogeology

The following sections provide a brief description of the regional and OU3-specific hydrogeology. More extensive information is presented in the RI report (LBG and Malcolm Pirnie 2012).

2.3.1. Regional Hydrogeology

The Passaic Formation generally forms a leaky multi-aquifer system that is several hundreds of feet thick. Groundwater movement is primarily through bedding plane fractures and steeply dipping interconnected fractures and dissolution channels (secondary permeability). A very limited amount of groundwater flows through the interstitial pore spaces between silt or sand particles because of compaction and cementation of the formation (primary permeability). Differences in permeability between layers resulting from variations in fracturing and weathering may account for many water-bearing units.

Groundwater in the Passaic Formation is often unconfined in the shallower, more weathered part of the aquifer; however, silt and clay derived from the weathering process typically fill fractures, thereby reducing permeability. This relatively low permeability

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surface zone reportedly extends 50 to 60 feet bgs (Michalski, 1990). Groundwater in the deeper portion of the Passaic Formation is generally confined as the lack of vertical fractures can create a confining effect with depth. Recharge is by leakage through fractures. The transmissivity of mudstone and siltstone units can range from 400 to 14,500 gallons per day per foot (Herman, 2001). Local and regional groundwater discharge boundaries include surface water bodies like Bound Brook. However, municipal pumping centers (water wells) account for most of the regional groundwater discharge.

The Passaic Formation contains an aquifer that is used as a source of potable water for some of the communities surrounding the former CDE facility (Figure 2-2). Numerous private, industrial, and municipal wells tap the formation, with reported pumping rates that range from a few to several hundred gallons per minute. Current groundwater extraction influences regional and local groundwater movement, and the variable historical configuration and pumping of municipal extraction wells exerted a dominant influence on historical groundwater movement at the former CDE facility. The following wellfields have been identified as having the most significant influence on that groundwater movement (details for these wellfields are presented in the RI report):

- Park Avenue Wellfield
- Tingley Lane Wellfield
- South Plainfield Wellfield
- Sprague Wellfield
- Spring Lake Wellfield

2.3.2. OU 3 Hydrogeology

The bedrock aquifer in OU3 is separated into three hydrogeologic units or water-bearing zones, identified as the “shallow”, “intermediate”, and “deep” water bearing zones. These zones were selected based on the location of monitoring ports to facilitate the preparation of depth-discrete potentiometric surface maps and CVOC distribution maps as it was important to select zones that contained one port at each well location. However, each of the zones selected does not necessarily coincide with where most of the fractures occurred. Each of these zones is hydraulically connected. It should be noted that there are numerous FLUTe™ ports between zones and some are deeper than the deep water bearing zone. The potentiometric surface data and concentration of CVOCs from these ports were also used in the overall interpretation of groundwater flow and CVOC distribution at and downgradient of the former CDE facility.

The shallow water bearing zone is unconfined and extends from the water table to a depth of approximately 120 feet bgs (bedrock). The water table fluctuates from the unconsolidated deposits due to seasonally high recharge and falls into the bedrock during seasonally low recharge and the effects of nearby pumping. Therefore, the groundwater encountered in the unconsolidated deposits is interpreted as part of the shallow unconfined bedrock aquifer. The shallow water bearing zone is hydraulically connected to surface water bodies, Cedar Creek, and Spring Lake. Groundwater to a depth of 120 feet bgs between MW-16 and ERT-3 has the potential to be hydraulically connected (discharging) to Bound Brook near the former CDE facility. The intermediate and deep water bearing zones are not hydraulically connected to surface water bodies. Even though the aquifer is highly fractured, there is some bedrock structure that produces localized anisotropic conditions. The portion of the groundwater between MW-16 and ERT-3 that cannot discharge to Bound Brook, due to the lack of vertical fractures, and the remaining portion of the water bearing zones will migrate to the north-northeast in an arc until it eventually reaches a downgradient receptor such as a municipal well.

The shallow water bearing zone is highly fractured. This is evidenced by the Theisian behavior of the aquifer (no fracture dewatering) in response to pumping during an Integrated Pumping Test. The intermediate and deep water bearing zones are also highly fractured; however, there is some evidence that the lack of vertical fractures in some locations create an anisotropy that influence groundwater movement and create a confining effect with depth (Michalski and Britton, 1997). The highly fractured nature of the bedrock was documented with hydraulic profiling and acoustic televiewer logging conducted as part of the borehole geophysics program. The hydraulically active fracture data were compiled and evaluated, and were used to generate the simulated fractured bedrock domain used in the FRACTRAN modeling. Each of these water bearing units is described below.

Shallow water bearing zone (water table to 120 feet bgs): The shallow water bearing zone is monitored by the uppermost port in each of the multi-port systems and the shallow bedrock wells constructed at the former CDE facility. An evaluation of current shallow bedrock groundwater levels compared to those collected during previous investigations indicate that current shallow bedrock aquifer water levels are approximately five feet higher than they were during the Foster Wheeler RI (FWENC, 2001). The water level variations are interpreted to be the result of historical groundwater pumping near Spring Lake, which was gradually reduced and ultimately stopped in 2003.

Intermediate water bearing zone (120 feet to 160 feet bgs): The intermediate water bearing zone marks the transition between the shallow and deep water bearing zones. This zone is monitored by the ports in each of the multi-port systems between 120 feet and 160 feet bgs. The fractures in the intermediate water bearing zone exhibit less in-filling with sediment and exhibit an increased permeability in individual fractures as compared to the shallow water bearing zone.

Deep water bearing zone (200 feet to 240 feet bgs): The deep water bearing zone exhibits an increased permeability, due to fractures being more open with less in-filling of material due to weathering. This zone is monitored by the ports in each multi-port system between 200 and 240 feet bgs.

2.3.3. Hydraulic Gradient and Groundwater Movement

The depth to water level was measured during three synoptic rounds (October 2009, March 2010, and July 2010). Each measurement was then subtracted from the surveyed elevation at the well to calculate a water level elevation in ft msl.

Groundwater elevations from shallow wells and the shallowest multi-level sampling port were used to characterize the Shallow water bearing zone collected in July 2010 (Figure 2-3). The data show that the potentiometric surface is generally affected by localized discharge to Bound Brook, Cedar Brook, and Spring Lake. Groundwater in the shallow water bearing zone moves away from the site in a radial pattern, moving north and east from the facility toward Bound Brook, and northwesterly toward the low-lying area at the confluence of Bound Brook and Cedar Brook. The relatively flat hydraulic gradient is anomalous and incongruent with the low hydraulic conductivity of the bedrock as characterized by an Integrated Pumping Test (IPT). Groundwater elevations in wells MW-19, MW-20, and MW-21 in the northwestern portion of OU3 have a significantly lower elevation reflecting the influence of the Park Avenue wellfield. To the northeast of the former CDE facility, immediately across Bound Brook, groundwater movement in the shallow water bearing zone is generally toward the west, with groundwater discharging to Bound Brook, Cedar Brook and Spring Lake.

Groundwater elevations from multi-level sampler ports between 120 and 160 feet bgs were used to characterize the intermediate water bearing zone collected in July 2010 (Figure 2-4). The generalized direction of groundwater movement is to the north with the gradient generally trending northwest near the former CDE facility before turning to the north-northeast as a result of the influence of local pumping centers. There is no groundwater-surface water interaction associated with the intermediate zone. The intermediate water bearing zone forms the transition between the shallow flow system that discharges to surface water, and the deeper, more regional, flow system that discharges to the pumping centers as shown by the hydrogeologic cross-section shown on Figure 2-6.

Groundwater elevations from multi-level sampler ports between 200 and 240 feet bgs were used to characterize the deep water bearing zone collected in July 2010 (Figure 2-5). The generalized direction of groundwater movement is to the north with the gradient generally trending northwest near the former CDE facility before turning to the north-northeast as a result of the influence of local pumping centers. A plot of the potentiometric surface indicates that the hydraulic gradient is more uniform in this zone, with no exhibited groundwater-surface water interaction.

A distinct, highly transmissive fracture zone was intersected by several boreholes during the RI. Most notably, this fracture zone underlies the overburden source area at MW-14S/D (the area where the highest concentration of the CVOCs were detected) at a depth of approximately 67 feet bgs, is present at MW-17 at a depth of approximately 200 feet bgs, and is present at MW-20 at a depth of approximately 300 feet bgs. At MW-14S/D beneath the overburden source area, the highly transmissive fracture zone marks a sharp decrease in both rock matrix and aqueous CVOC concentrations (discussed in Section 2). At downgradient areas, the location of the fracture zone is coincident with the highest concentration in the FLUTe™ wells. This suggests that the fracture zone limited vertical migration of the aqueous contaminant mass at the former CDE facility, and facilitated downgradient transport of the contaminant mass along a preferential (high transmissivity) pathway.

A hydrogeologic cross section is presented as Figure 2-6. The synoptic data were collected from each multi-level sampler port in July 2010, and show the horizontal and vertical components to groundwater movement in the study area. The vertical gradient varies across the study area and with depth. Groundwater elevations measured at multiple depths at MW-13, MW-16, ERT-3, and ERT-4 indicate upward hydraulic gradients at wells adjacent to Bound Brook, with lesser upward hydraulic gradients observed in wells at the former CDE facility, closer to the overburden source area at MW-14S/D. When compared to the corresponding stream gage measurements, the hydraulic head difference indicates the potential for groundwater discharge to Bound Brook. The upward vertical hydraulic gradients in the deep water bearing zone wells to the north of the former CDE facility (MW-20, MW-19) are likely related to anisotropic conditions and/or gradients created by groundwater extraction at the Park Avenue wellfield.

A comparison of historic groundwater elevations measured during the Foster Wheeler RI (2000) to the groundwater level measurements collected during the 2010-2011 RI show a marked change in groundwater elevations and the direction of groundwater movement in the shallow water bearing zone (Figure 2-2). Past groundwater elevations (2000) indicated that groundwater elevations were up to 5 feet lower and below the bottom of Bound Brook. Groundwater movement in the shallow water bearing zone at the former CDE facility was generally toward the northwest and beneath Bound Brook, with a potential for surface water in Bound Brook to recharge the aquifer. Current (2010-2011) conditions are different. Groundwater level measurements show shallow groundwater is potentially discharging into Bound Brook. Additionally, the groundwater elevations measured by Foster Wheeler (2000) were approximately 5 feet lower than those observed in the recent data (2010-2011). The Foster Wheeler data were collected under historic pumping conditions related to operation of the Middlesex Water Company's Spring Lake wellfield, which ceased pumping operations in 2003. The groundwater withdrawals from the Spring Lake wellfield in this area may have played a role in altering the regional hydrogeologic conditions, including a depression in local and regional groundwater elevations, alterations of the local gradients, and reversal of the local discharge/recharge potential between groundwater and surface water (Bound Brook). Today, hydrogeologic conditions at the former CDE facility are more influenced by the on-going groundwater withdrawals at the more distant Park Avenue wellfield.

3. Site Conceptual Model

3.1. Nature and Extent of Contamination

3.1.1. Contaminants of Concern

The RI and Baseline Human Health Risk Assessment (BHHRA) Reports (LBG/MPI 2012) identify the contaminants of concern (COCs) for groundwater by comparing the compounds detected to the potential ARARs. The list of COCs includes: VOCs, PCBs, Metals, Dioxin/Furans, Pesticides, and semi-volatile organic compounds (SVOCs). However, for the purposes of this TI evaluation, only VOCs (particularly chlorinated ethenes) are described in detail as their extent encompasses the area where all other COCs are present and their distribution in the rock matrix forms the basis for the waiver of ARARs.

3.1.2. VOCs

3.1.2.1. Non-Aqueous Phase Liquid (NAPL)

Following borehole drilling and prior to final FLUTe™ well construction, non-aqueous phase liquid (NAPL) reactive liners were installed in MW-14D, MW-15S, MW-15D, and MW-17 to test for the presence of NAPL. The reactive liners in MW-15S, MW-15D, and MW-17 did not indicate the presence of NAPL. Only the reactive liner in MW-14D indicated the presence of NAPL. Based on a visual inspection of the liner, the depth at which the NAPL entered the borehole appeared to be relatively shallow (approximately 70 feet bgs), near the top of the open bedrock interval. The reactive liner also showed that a small amount of NAPL pooled at the bottom of the borehole. NAPL was not observed at any other location during the RI.

3.1.2.2. Rock Matrix

As discussed in the RI Report, 465 split rock core samples were collected for analyses of select VOCs at four monitoring well locations (MW-14S, MW-14D, MW-16, MW-20). Sample locations were determined based on fracture distribution, with a minimum sample frequency of one sample for every two feet of core. Samples for VOC analyses were collected from fractured sections and surfaces as well as massive, un-fractured sections of core.

TCE was the most common VOC present in the rock matrix samples (345 detections), followed by cis-dichloroethene (cDCE; 96 detections), and tetrachloroethene (PCE; 27 detections). The rock matrix data were converted to equivalent matrix pore water concentrations to approximate the potential aqueous concentrations in the rock matrix at each sample interval. The equivalent matrix pore water concentrations are calculated using estimated and directly measured physical properties such as wet rock bulk density, dry rock bulk density, matrix porosity, soil-water partitioning coefficients, and organic carbon partitioning coefficients. The equivalent pore water concentrations of TCE and cDCE detected in rock matrix screening samples are shown on Figures 3-1 and 3-2, respectively. The following discussion of sampling results focuses on each of the rock matrix sample locations.

- <u>MW-14S/D</u>: VOCs were detected in approximately 70% of the rock matrix samples collected in the center of the former CDE facility from two borings (MW-14S and MW-14D). The equivalent pore water concentration of TCE in the rock matrix ranged from non-detections of less than the Practical Quantitation Limit (PQL) of 16 microgram per liter (µg/L) at depths of 90.5 to 94.9 feet bgs, and 99 to 106.8 feet bgs to 120,000 µg/L at 33.1 feet bgs. The concentration of cDCE in the rock matrix ranged from non-detections of less than the PQL of 390 µg/L at depths of 79.2 to 231.5 feet bgs to 330,000 µg/L at 33.1 feet bgs. PCE in the rock matrix ranged from non-detections or estimated concentrations less than the PQL of 31 µg/L at depths of 115.4 to 231.5 feet bgs to 130 µg/L at 75.95 feet bgs.

  The results indicate that the highest concentration of VOCs in the rock matrix at MW-14S/D was detected in the 23 to 75 feet bgs depth interval. The distribution of the results between 23 and 67 feet bgs indicates that contaminant mass has completely saturated the matrix blocks between fractures, indicative of very high historic aqueous concentrations, a dense fracture network, and sufficient time to completely diffuse into the matrix. The observed matrix block saturation and concentrations and the observed NAPL are consistent with a maturing CVOC aqueous mass that is approaching equilibrium conditions, as identified in the conceptual model.

- <u>MW-16</u>: CVOCs were detected in approximately 90% of the samples collected from one boring (MW-16) near the northern boundary of the former CDE facility. The equivalent pore water concentration of TCE in the rock matrix ranged from non-detections of less than the PQL of 3.4 µg/L at depths of 214.4 to 224.7 feet bgs to 7,800 µg/L at 46.7 feet bgs. The concentration of cDCE in the rock matrix ranged from non-detections of less than the PQL of 520 µg/L at depths of 202.1 to 251.6 feet bgs to 4,500 µg/L at 175.95 feet bgs. PCE in the rock matrix was detected in just two samples, at concentrations less than 30 µg/L at depths of 125.55 and 128.45 feet bgs.

  The results indicate that CVOC mass was detected throughout the entire cored interval. The highest concentration of VOC mass at MW-16 was detected in the 50 to 150 feet bgs depth interval (intermediate water bearing zone). The distribution of the results between 50 and 150 feet bgs indicate that contaminant mass has saturated matrix blocks between fractures. Between 150 and 200 feet bgs, the rock matrix concentrations decrease steadily, and the distribution of mass becomes more prominent. This suggests that contaminant mass is present in fewer fractures, and at decreasing concentrations.

- MW-20: CVOCs were detected in approximately 80% of the samples collected from one boring (MW-20) adjacent to Spring Lake. The equivalent pore water concentration of TCE in the rock matrix ranged from non-detections of less than the PQL of 14 µg/L at depths of 28 to 35.2 feet bgs to 1,100 µg/L at 295.6 feet bgs. The concentration of cDCE in the rock matrix was detected in just five samples, at estimated concentrations (data flagged with "J" qualifiers) of less than 63 µg/ L at depths of 70.8 to 74.65 feet bgs, at 76.9 feet bgs, and at 94.3 feet bgs. PCE in the rock matrix was not detected at MW-20.

  The results indicate that CVOC mass was detected throughout the entire cored interval. The largest proportion of CVOC mass was detected from 220 to 350 feet bgs depth interval (deep water bearing zone). The distribution of results between 28 and 220 feet bgs indicate presence of contaminant mass "halos" around discrete fractures (at approximately 85, 135, and 155 feet bgs), and that the concentrations in the rock matrix are relatively low. The results also indicate that matrix block saturation has occurred between 220 and 250 feet bgs and between 255 and 355 feet bgs. The concentrations in these zones are relatively low as compared to those encountered in MW-14 and MW-16, but the consistent elevated results are indicative of matrix block saturation. These zones probably represent dense fracture zones that are in direct or indirect communication with impacted groundwater. The interval between 255 feet bgs and 355 feet bgs is believed to be the same fracture zone identified at MW-14S/D installed at the overburden source area at MW-14S/D. The distribution of results at MW-20 indicate that the contaminant mass has diffused into rock to depths of 400 feet bgs, and that the greatest impact is concentrated near a fracture zone encountered at approximately 300 feet bgs. This fracture zone facilitates both groundwater movement and contaminant mass transport.

3.1.2.3. Groundwater

As discussed, the fractured rock aquifer was divided into three water bearing zones (Shallow, Intermediate, and Deep) to assist in the development of the site conceptual model and to describe the hydrogeology and distribution of contamination. The shallow water bearing zone represents the conditions at or near the top of bedrock, and is generally located between 20 feet and 60 feet bgs. The intermediate water bearing zone is generally located between 120 feet and 160 feet bgs across OU3. The deep water bearing zone is generally located between 200 feet and 240 feet bgs across OU3.

- Shallow Groundwater: The highest concentration of CVOCs was detected in the bedrock beneath the overburden source area at MW-14S/D near the center of the former CDE facility, at depths between 23 and 75 feet bgs, with concentrations falling off sharply at depths greater than 75 feet bgs. Figure 3-3 shows the areal distribution of CVOCs in the shallow groundwater. The resultant VOC mass in the shallow bedrock has moved to the northwest, consistent with both the observed shallow groundwater gradient, and the historic gradient reported in the previous shallow bedrock investigation. The shallow water bearing zone impacts are generally limited to the area south of Bound Brook, as the surface water body acts as a boundary to shallow groundwater movement. However, elevated concentrations of CVOCs in the shallow water bearing zone were detected north of Bound Brook in ERT-4, MW-20, and MW-21. The elevated results at these locations suggest vertical mass transport along steeply dipping fractures.

- Intermediate Groundwater: Figure 3-4 shows the areal distribution of CVOCs in the intermediate groundwater. The groundwater data show a more northwesterly distribution of contaminants near the former CDE facility, with a northeastward arching path of travel towards the capture zone of the currently operating Park Avenue wellfield to the north.

- Deep Groundwater: Figure 3-5 shows the areal distribution of CVOCs in the deep groundwater. As with the distribution of aqueous mass described in the intermediate water bearing zone, the groundwater data show a more northwesterly distribution of contaminants near the former CDE facility, with a northeastward arching path of travel towards the capture zone of the currently operating Park Avenue wellfield.

3.2. Occurrence and Movement of Groundwater in Fractured Sedimentary Rock

Fractured sedimentary rock can be very difficult to characterize as it is highly heterogeneous and often anisotropic. The nature of the hydrogeologic system is dependent on a variety of factors, including rock matrix porosity and permeability, as well as fracture orientation, density and size.

Groundwater in fractured sedimentary rock occurs in the pore spaces or matrix of the rock (primary porosity), and in fractures of the rock (secondary porosity). This type of bedrock can be described as a “dual porosity” hydrogeologic system, where the primary porosity is the porosity of the rock matrix (pore spaces) and the secondary porosity is the porosity of the bedrock fractures. The primary porosity of the rock matrix is relatively high, typically between 5% and 20%, because a large volume of water can be stored in the pore spaces of the bedrock. Conversely, the secondary porosity of the rock fractures is relatively low, typically between 0.1% and 0.001%, because a much smaller amount of water can be stored in the fractures. The primary and secondary porosity of a dual porosity hydrogeologic system only refers to the total amount of water stored in the rock matrix (pore spaces) and fractures. It does not have any correlation to movement of water through the rock matrix or fractures.

The degree of interconnectedness of the pore spaces within the rock matrix, termed primary permeability, affects the degree to which groundwater can move through the pore spaces or rock matrix. The primary permeability of the rock matrix is very low because even though a large volume of water is stored in the pore spaces of the rock matrix, the interconnectivity of the pore spaces of the rock matrix is very low due to the small grain size of the silt, the small pore spaces of the rock matrix, and the fact that a portion of the pore spaces of the matrix has been filled with material that cements the individual silt grains together to form the consolidated bedrock. The degree of interconnectedness of the individual fractures, termed secondary permeability (also known as bulk hydraulic conductivity in fractured bedrock aquifers), affects the degree to which groundwater can move through the fracture network. The secondary permeability of bedrock fractures is often much higher than the primary permeability of the rock matrix.

Therefore, the bedrock matrix has a high porosity (ability to store water) but a low permeability (ability to transmit the stored water). Conversely, the bedrock fractures have a low porosity (ability to store water) but a high permeability (ability to transmit water).

3.2.1. DNAPL Contamination in Fractured Sedimentary Rock

DNAPLs are among the most persistent contaminants in groundwater. When released into the environment, a DNAPL will flow downward through the unsaturated zone. The DNAPL will also flow downward through saturated porous media because it’s denser than water. However, DNAPLs are non-wetting fluids and they have a very high surface tension, both of which affect the flow properties of the fluid and can lead to pooling.

Upon reaching the top of fractured sedimentary rock, the DNAPL will pool in areas of low permeability and they will continue to migrate downward through the more transmissive fracture zones. The typically very low fracture porosity allows the DNAPL to migrate laterally and vertically great distances, far more than it would migrate in an equal volume of a porous medium (Feenstra and Cherry, 1988). DNAPL typically penetrates the fracture network, working into ever smaller openings, creating pools, fingers and disconnected globules of residual contamination. With time, the DNAPL will dissolve into groundwater and move as aqueous mass, which is then subject to dispersion, diffusion, sorption, and degradation (abiotic and biotic) processes.

Several groundwater studies have been conducted to understand the dynamic equilibrium between the advective fracture flow of aqueous mass and the diffusion of aqueous mass into the low permeability matrix. These studies show that the diffusion process is driven by the concentration gradient between the aqueous mass in the fracture and the matrix pore water.

In the early stages of aqueous mass movement in fractures, diffusion into the matrix can slow the advance of the aqueous mass in the fractures. In this stage, the aqueous mass does not move as quickly as groundwater that can be characterized by advective flow velocities because diffusion, sorption, and degradation are attenuating the leading edge of the aqueous mass. The aqueous mass is dispersed in the fracture network, which provides a large total surface area for attenuation processes. Early in the matrix diffusion process, most of the diffused mass occurs as ‘halos’ around discrete fractures indicating that the mass has penetrated only a short distance into the bedrock (Parker et al., 1994).

As the plume matures, the rock matrix and aqueous fracture concentrations approach equilibrium. In addition, the advance of the aqueous mass in fractures slows and even potentially stops as the aqueous mass concentration gradients in the fractures and matrix reach a dynamic equilibrium. Dynamic equilibrium is generally achieved after a significant time period (~50 years). In cases with large DNAPL releases over a period of time (as evidenced at the CDE Site), the high aqueous mass concentrations can drive the matrix diffusion process beyond the contaminant halo, to where the aqueous mass penetrates more than a few millimeters and totally saturates the matrix block.. This effect more commonly occurs in source areas, where aqueous mass concentrations are highest and the residence time is the longest.

After a significant period (50 years) of time in the fractured bedrock environment, contaminant mass (i.e., DNAPL and or high concentrations of dissolved-phase mass) has been driven into the rock matrix by diffusion and aqueous-phase mass has been transported down gradient from the overburden source area (i.e., MW-14S/D). The aqueous-phase mass concentrations in the fractures will be lower than the mass concentrations driven into the rock matrix. At this point, the process of matrix diffusion will reverse (back diffusion), releasing the mass in the rock matrix (pore water) back to the aqueous-phase in the fractures over a very long period of time (usually in multi-century timeframe). In addition, the distal portions of aqueous-phase mass will be stabilized because of attenuating processes (diffusion-driven mass transfer into the matrix, sorption, and biotic and abiotic degradation) that can significantly slow or stop the advance of the leading edge of the contaminant mass. However, as a result of on-going back diffusion, these types of impacted aquifers cannot be restored to their highest beneficial use in a reasonable timeframe and at a reasonable cost.

3.2.2. Fate and Transport of Chlorinated Ethenes at OU-3

DNAPLs are denser than water, typically less viscous than water, often resulting in rapid rates of subsurface migration. Additionally, these compounds typically have low K_{oc} values (the affinity of a compound to adsorb to soil and is dependent on the amount of organic carbon present in the system), indicating a low degree of sorption. As discussed above, preferential DNAPL migration through the larger aperture fractures subsequently establishes an aqueous concentration gradient driving mass into the porous matrix by diffusion (Parker, 2007).

Shallow rock matrix TCE pore water equivalent concentrations from MW-14S are approximately 10% of the solubility limit, and the concentrations in the rock matrix generally exceed the aqueous concentrations from adjacent monitoring ports. The deep rock matrix pore water equivalent concentrations from MW-14D are much lower, but still exceed the aqueous concentrations from adjacent monitoring ports. The cDCE pore water equivalent concentrations at MW-14S show a similar relationship; however, the deep bedrock matrix concentrations data (MW-14D) indicate smaller relative proportion of cDCE in the rock matrix at depth. The relatively high concentration of CVOC in the pore water equivalent, and the saturation of CVOC throughout the matrix block, suggest that the aqueous mass in this location has or is approaching maturity. Downgradient transport of CVOC is facilitated by dissolved aqueous mass moving through the fracture network. Diffusion into the rock matrix occurs continuously wherever the concentration gradients of dissolved CVOC are sufficient to drive the process. Advective transport of dissolved CVOC mass through the fracture network is the main process behind the downgradient advance of the leading edge of the aqueous mass; however, there are other processes at work which act to slow or retard the advance of the leading edge of aqueous mass. Toward the distal end of the aqueous mass (MW-20), the lack of sufficiently high concentration gradients and time for diffusion to occur is indicated by relatively high aqueous phase concentrations compared to the rock matrix pore water equivalent concentration.

3.2.3. Summary

Groundwater flow in the Passaic Formation occurs primarily through the fracture network. The network is composed of bedding parallel to sub-parallel fractures with steeply dipping joint sets and is highly conductive and interconnected, allowing for the horizontal and vertical movement of groundwater. The average fracture aperture size is 83 microns, or slightly smaller than the thickness of a human hair. The extremely small size of the apertures, and an average fracture frequency of 0.9 fractures per every linear foot, gives the fracture network a relatively low porosity (2.1 x 10^{-5} ft^3/ft^3) as compared to the porosity of the matrix rock (0.1 ft^3/ft^3). However, the fracture frequency, volume, and interconnectedness give the network a moderate bulk hydraulic conductivity (2.2 to 5.5 ft/day) and allows for both vertical and horizontal groundwater flow.

The aquifer is divided into three hydrogeologically connected units (for discussion purposes): the shallow, intermediate, and deep water bearing zones. The shallow water bearing zone is unconfined and extends from ground surface to a depth of approximately 120 feet bgs (unconsolidated materials and bedrock). The current phreatic surface in shallow bedrock (water levels recorded in the shallow bedrock aquifer) is above the top of bedrock, and within the unconsolidated deposits, and has risen approximately five feet since the initial groundwater investigation was conducted by Foster Wheeler in 2000. There is some evidence that the lack of vertical fractures in some locations create anisotropy that influence groundwater movement and create a confining effect with depth (Michalski and Britton, 1997) The fracture network exerts an increasing control over groundwater movement below about 250 feet bgs, due to a decrease in the frequency of fractures.

Water level measurements taken during the RI indicate that the water table measured in the shallow water bearing zone is generally controlled by localized discharge to Bound Brook, Cedar Brook, and Spring Lake. Groundwater in the shallow water bearing zone discharges to Bound Brook, Cedar Brook, and Spring Lake and moves north and east from the former CDE facility toward Bound Brook and northwest toward the low-lying area at the confluence of Bound Brook and Cedar Brook. Groundwater movement in both the intermediate and deep water bearing zones is primarily to the northwest at the former CDE facility and arcs to the north and northeast with increased proximity to the Park Avenue wellfield.

The highly conductive fracture network allows for the vertical and horizontal advection of groundwater and aqueous mass. Because the fracture network is so pervasive, it provides a relatively large surface area for the CVOCs to sorb onto and then diffuse into the rock matrix. The pore volume of the rock matrix is nearly two orders of magnitude larger than the fracture network, allowing it to hold the majority of the contaminant mass. Once the aqueous mass has diffused into the rock, it is left nearly immobile because of the low hydraulic conductivity of the rock matrix. In addition to sorption and diffusion, microbiological analyses indicate that the degradation of CVOCs is occurring, which contributes to the retardation of the advance rate of the leading edge of aqueous mass.

The aqueous mass migration has also been influenced by ongoing withdrawals at the Park Avenue and Sprague wellfields, by intermittent pumping at Spring Lake which took place between 1964 and 2003, intermittent pumping at the Tingley Lane wellfield which took place between 1954 and 2010, and by historic pumping at the South Plainfield Wellfield which reportedly took place between approximately 1952 and 1969. Although the general direction of groundwater movement beneath the former CDE facility is to the northwest, the pumping centers to the north and east of the former CDE facility redirected the groundwater movement and mass transport. Today, groundwater extraction at the Park Avenue and Sprague wellfields is the dominant hydraulic influence on the regional and local hydrogeology.

A distinct, highly transmissive fracture zone was intersected by several boreholes during the investigation, which facilitated the down gradient transport of aqueous mass along a preferential (high transmissivity) pathway. While pumping at active wellfields was occurring, the downward vertical component of the groundwater gradient was higher, thereby increasing the downward movement of the contaminant mass. This fracture zone is capable of conducting the aqueous mass down gradient, toward the nearest active pumping wells to the north - the Park Avenue wellfield.

The influence of the various pumping centers in the area created a highly variable flow field within the fractured rock aquifer. While the direction of groundwater movement may have shifted locally during pumping at the South Plainfield and Spring Lake wellfields, the general regional gradient was toward the north, influenced by the most productive wellfield in the area (Park Avenue). In addition, periods of heavy groundwater usage would have lowered regional groundwater levels, reversing the head relationships between groundwater and surface water.

These changes in conditions are likely to cause advective redistribution of the aqueous mass. In areas where the concentration of the aqueous mass in fractures is greater than that in the adjacent matrix pore water, diffusion into the rock is occurring and attenuating the leading edge of the aqueous mass. Furthermore, back diffusion out of the matrix (pore water) is occurring in areas where the concentration gradient between the rock matrix and the aqueous phase in fractures supports the process, which would contribute to ongoing groundwater contamination over a very long period of time (usually in a multi-century timeframe). As a result, the contaminated aquifer cannot be restored such that it meets ARARs, allowing use as a potable water supply without treatment at the wellhead, in a reasonable timeframe and at a reasonable cost.

4. Evaluation of Site Restoration Potential

Under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), groundwater restoration cleanup levels are established by chemical-specific ARARs. To evaluate the restoration potential for groundwater at OU3, the impact of the critical limitations posed by the nature of the contaminant distribution (i.e., rock matrix) and its extent (OU3 groundwater plume) on the conceptual effectiveness of currently available potential remedial technologies was evaluated as part of this "up-front" TIER. As a basis for the evaluation, a fate and transport model was developed and utilized to evaluate the effects of the implementation of remedial efforts at the former CDE facility on OU3 groundwater.

4.1. Fate and Transport Modeling

In support of the RI, contaminant (CVOC) fate and transport modeling was conducted to evaluate the extent of contaminant migration in the bedrock groundwater and the impact of potential bedrock treatment remedies. The modeling was conducted and reviewed by Steven Chapman, Dr. Beth Parker, and Dr. John Cherry of the University of Guelph. The results of the modeling are presented in the *Draft Report on Discrete Fracture Network (DFN) Contaminant Transport Modeling, Cornell-Dubilier Electronics Superfund Site – OU3 Groundwater*, dated June 2011 (DFN Modeling Report) (Chapman 2011), which is included in Appendix A, and briefly summarized herein.

Integrated pumping tests at the CDE site show that the groundwater flow system in the highly fractured bedrock can be reasonably simulated as an equivalent porous media (EPM). However, evaluation of contaminant fate and transport must consider effects of matrix diffusion on contaminant behavior in discretely fractured rock systems. While fractures provide the dominant pathways for groundwater flow, the large rock matrix porosity represents the bulk of the contaminant mass storage capacity. Thus, diffusion of contaminants into the rock matrix in this dual porosity system, as well as sorption within the matrix and potentially contaminant degradation, is expected to have a strong influence on contaminant behavior and remedial efficacy.

4.1.1. Model Approach

The modeling approach applied at the CDE site involved application of the MODFLOW EPM model to simulate the groundwater flow system to obtain overall bulk flow characteristics (i.e., hydraulic gradients, bulk hydraulic conductivity and groundwater fluxes) and then the discrete fracture network (DFN) model FRACTRAN was used to simulate contaminant fate and transport. The purpose of the DFN transport simulations is to represent groundwater flow and contaminant transport in fractured porous media incorporating relevant processes of rapid groundwater flow in fractures and contaminant diffusion into and out of the rock matrix.

The FRACTRAN DFN simulations were conducted for TCE assuming no degradation. Data from the site suggest transformation of TCE to cis-DCE occurs, but it is unknown whether further dechlorination occurs since groundwater data show little vinyl chloride (VC) presence. The model domain for the site simulations was a vertical cross-section 1000 meters (m) long and 150 m high. The fracture network was selected after producing several realizations of randomly generated fracture networks and adjusting the key fracture network statistics including mean fracture aperture and variance, fracture density, and fracture length ranges to provide an overall horizontal bulk hydraulic conductivity within a target range based on the field data (e.g., FLUTe™ liner test data and pump test data) and MODFLOW EPM flow model results. The vertical head component was set to match the apparent plume deepening with depth based on the rock core VOC results. The significant mass in the bedrock was positioned within the upper portion of the model domain consistent with the apparently limited DNAPL penetration beneath the overburden source area at MW-14S/D.

4.1.2. Model Results

The results of the FRACTRAN DFN transport simulations, which take into account the current pumping effects from the Middlesex Water Company (MWC) supply wells, showed that contaminant migration in the fracture network is much slower than groundwater flow rates in fractures, due to attenuation processes including diffusion of mass from fractures to the rock matrix. However, by 10 years after the release, the simulation results show contamination has already reached the model boundary at 1000 m and, by 50 years, contamination occurs throughout the model domain.

4.1.3. Future Projections of Site Restoration Potential

For future projections, two scenarios were assumed: (1) continued input at 10% of solubility, and (2) complete removal of the source input term. The results show little impact of complete removal of mass input on persistence of the downgradient plume, which may be expected given that the majority of the contaminant mass exists in the rock matrix. While some minor improvements in groundwater quality within the plume are achieved from the removal of significant mass in the bedrock, the time to achieve such benefits are extremely long and concentrations still remain elevated for very long time periods (i.e., on the order of several hundred years). This is consistent with findings at other fractured bedrock sites studied by researchers at the University of Waterloo, the University of Guelph, Queens College, and Colorado State University.

Actual bedrock conditions at the CDE site are likely in between the two scenarios given ongoing remedial activities at OU2 to remove contaminated overburden materials. While these FRACTRAN DFN simulations do not incorporate a sufficiently large domain to capture the full simulated plume extent, the expectation is that the rate of plume front migration (beyond the extent of the model domain) would be very slow at present time due to effects of matrix diffusion. These simulations also suggest efforts to completely remove the significant mass in the bedrock beneath MW-14S/D would have a negligible impact on down gradient rock matrix and groundwater conditions over the next several hundred years.

4.2. Potentially Applicable Technologies

The following programs/information resources were used to identify and review new technologies for remediation of CVOCs in groundwater in alluvial aquifers that may potentially be applicable to the OU3 bedrock groundwater:

- United States Environmental Protection Agency (USEPA) Superfund Technology Innovation Program (TIP) (formerly the Technology Innovation Office (TIO);
- USEPA Cleanup Information (CLU-IN) Website (www.clu-in.org);
- Federal Remediation Technologies Roundtable;
- Environmental Security Technology Certification Program (ESTCP);
- Strategic Environmental Research and Development Program (SERDP);
- Interstate Technology and Regulatory Council (ITRC);
- Air Force Center for Engineering and the Environment (AFCEE);
- Naval Facilities Engineering Command (NAVFAC);
- National Groundwater Association – Groundwater On-line Database;
- University of New Hampshire Bedrock Bioremediation Center; and
- U.S. Geological Survey (USGS) Toxic Substances Hydrology Program.

In addition, the following publications were reviewed:

- Federal Remediation Technologies Roundtable. 2009. Treatment Technologies Screening Matrix.
- Geosyntec Consultants. 2007. Final Report, Bioaugmentation Pilot Study, Former Naval Air Warfare Center, West Trenton, New Jersey. June 2007.
- Interstate Technology Regulatory Council (ITRC). 2008. In Situ Bioremediation of Chlorinated Ethene DNAPL Source Zones. June 2008.
- LaChance, John (TerrraTherm) and Pierre Lacombe (USGS). 2009. Thermal Treatment of DNAPL in Fractured Bedrock Using Thermal Conduction Heating.

Presentation for 2009 Fractured Rock Technology Seminar and Guided Site Tour, Naval Air Warfare Center, West Trenton, New Jersey. June 2009.

- McDade, James M., Travis McGuire, Charles Newell. 2005. Analysis of DNAPL Source-Depletion Costs at 36 Field Sites. Spring 2005.

- National Groundwater Association. Fractured Rock: State of the Science and Measuring Success in Remediation. September 2005.

- National Research Council (NRC). 2005. Contaminants in the Subsurface, Source Zone Assessment and Remediation.

- Sale, Tom, Charles Newell, Hans Stroo, Robert Hinchee, and Paul Johnson (ESTCP). 2008. Frequently Asked Questions Regarding Management of Chlorinated Solvents in Soils and Groundwater. July 2008.

- Strategic Environmental Research and Development Program. 2007. Project Fact Sheet: A Comparison of Pump-and-Treat Natural Attenuation, and Enhanced Biodegradation to Remediate Chlorinated Ethene-Contaminated Fractured Rock Aquifers, Naval Air Warfare Center, West Trenton, New Jersey. October 2007.

- United States Environmental Protection Agency. 2005. Steam Enhanced Remediation Research for DNAPL in Fractured Rock, Loring Air Force Base, Limestone, Maine. National Risk Management Research Laboratory, Cincinnati, Ohio. August 2005.

Remedial strategies presented in these information sources focused primarily on potential methods for in-situ source treatment in alluvial aquifers. Treatment technologies employed included:

- In-situ enhanced bioremediation (ISB);

- In-situ chemical oxidation (ISCO);

- Monitored natural attenuation (MNA); and

- In-situ thermal remediation (ISTR).

Technologies reviewed and/or presented in the majority of the reviewed sources focused on source treatment as a means to reduce contaminant concentrations and shorten remedial timeframes. Much of the available literature focuses on treatment of CVOCs in unconsolidated media; however, limited information pertaining to fractured bedrock sites was available. This information included compilations of work at several sites and two case studies. In general, these studies yielded the following information:

1. In all cases, the remedial methods focused on source treatment or containment.

2. ISCO, primarily with permanganate as the oxidant, has been used widely as the preferred source remediation technology at fractured bedrock sites where groundwater was contaminated with chlorinated VOCs. However, the use of ISCO did not result in the achievement of ARARs at any of the sites reviewed.

3. Containment technologies were the most commonly used remedial technology – particularly for plume containment at site with both bedrock and unconsolidated aquifers. However, several sources, including a USEPA study of 28 sites at which groundwater containment remedies have been implemented, found that, while most sites have met plume containment goals, only a handful of sites have met their aquifer restoration goals (USEPA, 1999). These data indicate that, while containment remedies are viable remedies to eliminate potential exposure pathways, they are not effective source treatment technologies enabling attainment of ARARs. This is particularly true at fractured sedimentary bedrock sites where matrix diffusion plays a major role in contaminant persistence.

4. ISB and, more recently ISTR, have been implemented as source treatment technologies at bedrock sites at the small-scale pilot study level only.

5. Delivery and distribution of injected materials, and treatment of contaminants in microfractures, low flow zones, and the rock matrix presented the biggest obstacle to the success of source treatment technologies that rely on the injection of amendment or oxidants in fractured bedrock. To date, these challenges have not been overcome and the use of these technologies has not resulted in the achievement of groundwater standards at fractured bedrock sites.

6. Release of contaminants stored in the bedrock matrix are expected to sustain contaminant discharge for extended periods of time (i.e., hundreds of years), even where aqueous source treatment activities have been implemented.

7. In almost all cases, it was stipulated that achievement of ARARs (i.e., USEPA Maximum Contaminant Limits [MCLs]) was likely not possible within a reasonable timeframe, with reasonable timeframe loosely defined as 200 years.

Information on the following three relevant case studies was also reviewed:

1. ISTR pilot at the former Loring Air Force Base (AFB) in Limestone, Maine;

2. ISB and ISTR pilots at the former Naval Air Warfare Center (NAWC) in West Trenton, New Jersey; and

3. Full-scale ISCO Corrective Measures at Watervliet Arsenal (WVA) in Watervliet, New York.

The results of these case studies are summarized below:

4.2.1. Loring Air Force Base

Loring AFB was added to the Superfund National Priorities List in 1990. Subsequent investigations showed that chlorinated VOCs were present in the bedrock groundwater beneath a former quarry. The ROD, signed in 1999, recognized that it was impractical to restore groundwater in fractured rock to drinking water standards. However, an agreement was made between the United States Air Force (USAF) and the USEPA Region 1 to use the quarry to conduct a research project to further the development of remediation technologies in fractured rock with the hope of recovering contaminant mass to reduce the timeframe for natural attenuation of the remaining contaminants. Steam enhanced remediation (SER) was chosen as the preferred remedial technology for the site.

The results of the study showed that the amount of energy that could be injected during the limited-time project was low and that the target zone for treatment could not be completely heated. Despite the limited heating that occurred, effluent vapor and water samples showed that some contaminants were removed, likely as a result of air stripping from fracture surfaces. However, the amount of contaminants removed was limited and had no discernable impact on groundwater concentrations. The study concluded that steam injection may not be the best method for remediation for highly complex low permeability fractured rock sites and that extremely long injection times would likely be necessary for any full scale operations. However, even with long injection times, heat losses would likely limit the ability to heat the entire target zone.

4.2.2. NAWC West Trenton

NAWC has been the subject of an active remediation program since 1993. Historical releases of chlorinated solvents at the site led to the presence of elevated concentrations of CVOCs in the bedrock groundwater. The current remedial system is based on pumping and treatment of impacted groundwater and has been operating since 1997. The primary purpose of the system was to contain the CVOC plume and prevent off-site migration. The concentrations of CVOC in groundwater at monitoring points have generally decreased in the period from 1997 to the present, but have remained greater than groundwater quality standards. The sedimentary fractured bedrock at the site that is similar to the bedrock at the former CDE facility has been extensively characterized using similar methods to those used at CDE, including rock matrix VOC characterization, borehole geophysical testing by the USGS, and multi-level groundwater monitoring well installation.

ISB Pilot Study

A bioaugmentation pilot study was conducted to evaluate whether source treatment through ISB could potentially accelerate the shut-down of the groundwater extraction system. The study included the injection of an electron donor (emulsified soybean oil) and a culture containing TCE-degrading bacteria (KB-1®) into two well pairs. The total size of the treatment area was approximately 9,000 square feet and extended 120 feet below ground surface. Extracted water from one well was dosed with the injection materials and injected into its paired well within the test plot area. The results of the pilot showed that TCE concentrations in the test area were reduced. However, back-diffusion from the matrix resulted in contaminant rebound, which necessitated additional donor injections. These data indicate that the treatment method did not address CVOC contamination in the rock matrix, which will continue to act as a continuing source of aqueous groundwater contamination that will require long-term operation of the site-wide groundwater extraction remedy.

ISTR Pilot Study

An additional pilot study utilizing ISTR of the fractured bedrock by thermal conduction heating (TCH) was conducted in a 36-foot by 36-foot test area in another portion of the NAWC plume. The depth of treatment was approximately 55 feet below ground surface (bgs). The pilot test area includes 15 heater wells, three groundwater sampling/temperature monitoring wells, and five additional temperature monitoring points. The heater wells also served as vapor recovery points and the entire test area was covered with a concrete pad to facilitate the collection of vapors. Based on discussions with NAVFAC personnel, the initial results of the pilot were promising. Target temperatures were reached in the majority of the treatment area within approximately four months, with the exception of some zones where fractures appeared to inhibit heating, and significant VOC mass was recovered through the vapor extraction wells. The total subcontractor cost for the pilot was $500,000 and the approximate 4-month energy cost was $85,000, which included a dedicated electricity supply connection. Based on PID readings in the vapor effluent, the pilot removed approximately 275 pounds of VOCs, which is equivalent to only approximately 0.5 percent of the total estimated VOC mass at OU3.

4.2.3. Watervliet Arsenal

The WVA conducted a Corrective Measures (CM) program for the bedrock groundwater at Building 40 of the WVA, which is located in the City of Watervliet, New York. The goal of the CM program was to treat CVOCs present in the bedrock groundwater and shale bedrock matrix within an approximately 400 foot by 300 foot area at the WVA property boundary. The treatment program included injections of sodium permanganate (permanganate) and subsequent contaminant mass flux discharge monitoring to evaluate remedial efficacy.

The goal of the CM program was to reduce the concentration of hazardous constituents in groundwater migrating from the site to New York State and federal groundwater standards. However, given the presence of DNAPL in the fractured rock at the site, it was recognized by all parties (including State and Federal regulators) that the standards might not be achievable using currently available technologies. Accordingly, the CM program was subject to the following Performance Criteria:

1. Permanganate Distribution: The permanganate must be well distributed to and within the boundary monitoring wells within one year after the initiation of full scale injections.

2. Permanganate Residence Time: The permanganate must persist for at least 30 days after injection in the boundary monitoring wells within two years after the initiation of full scale injections.

The corrective measures were initiated in September 2004 with injections into a limited number of injection wells upgradient of Building 40. Full scale injections into all injection wells were initiated in August 2005. The maximum permanganate distribution in the compliance boundary monitoring wells was achieved during the first full-scale injection event in August 2005. Beginning with the November 2005 injection event, and in subsequent injection events, injection well clogging limited the amount and/or rate of oxidant that could be delivered to injection wells.

The results of the CM program were:

1. The permanganate injections did not decrease groundwater CVOC concentrations or the contaminant mass flux discharge at the compliance boundary. Rock matrix CVOC pore water concentrations also did not decrease after the two years of injections.

2. Persistent clogging problems indicated that a large portion of the injected permanganate mass was being oxidized to insoluble precipitates through interaction with the rock matrix, specifically the reduced sulfur (i.e., pyrite), present in the rock. This interaction with the rock greatly limited the effectiveness of the permanganate injections. Rock core, water level, pressure, and temperature monitoring also showed that the injections influenced only a portion of the treatment area, despite previous comprehensive geophysical, hydrogeologic, and rock matrix characterizations that indicated that injections should be capable of contacting all of the treatment area.

3. The CM program failed to achieve the CM Performance Criteria and, therefore, did not achieve the overall goal of reduction of CVOC concentrations in groundwater to state or federal standards.

Based on these data, and the lack of any other potentially effective remedial technology, it was determined that achievement of the groundwater standards at the site was not technically feasible using currently available technologies

4.3. Conclusions

Contaminant fate and transport modeling results indicate that treatment of bedrock limited to the area beneath the overburden source area (i.e., at MW-14S/D) would have negligible impact on the remainder of the downgradient plume and would not result in meeting ARARs since the bedrock matrix itself is the source of the ongoing exceedence of ARARs. Therefore, to be potentially capable of meeting ARARs, a remedial technology would have to be applied over the *entire* OU3 area where bedrock matrix contamination contributes to ongoing exceedences of ARARs. This would encompass an area of more than 150 acres and depths of more than 250 feet below ground surface. The implementation of any of in-situ remediation technology over such an area is not practicable.

As an example, the conceptual cost of using of ISTR, which is *potentially* capable of treating the chlorinated ethenes present in the bedrock groundwater and rock matrix, was evaluated to provide a basis for the impracticability of groundwater restoration. The use of ISTR under such a scenario would involve:

1. Treatment of the entire OU3 area would require physical access to, and the installation of treatment infrastructure at, *all* of the properties located within the area of bedrock matrix contamination. There are hundreds of properties in this suburban environ.

2. Spacing of the ISTR wells would have to be relatively close (on the order of 20 to 50 feet) to overcome the large heat loss potential resulting from the presence of both known and unknown fracture pathways.

3. ISTR, where successful, results in the volatilization of contaminants from the groundwater to the soil vapor. These vapors must be collected at the ground surface through a vapor extraction system. If they are not collected, they may present a vapor intrusion risk to surface structures in the area. Therefore, vapor extraction infrastructure would have to be constructed over the entire ISTR area, which includes numerous residences.

The NAWC pilot treated a volume of approximately 2,600 cubic yards (cy) at a cost of $585,000. This equates to approximately $2,100 per pound of VOC removed, or approximately $225 per cy treated. Given that the approximate total volume of the rock matrix-impacted area of OU3 is approximately 60,000,000 cy the resulting treatment cost would be in excess of $1 billion. Likewise, a study conducted by McDade et. al., found that the average cost of ISTR is approximately $100 per cy in unconsolidated materials.

As demonstrated by the NAWC pilot, these costs would be expected to be higher in bedrock due to the close heater well spacing that would be required.

4.4. Off-Site Sources

As discussed in the RI Report, there is evidence of other sources of groundwater contamination in the area of the Site that contribute to the exceedence of ARARs in and downgradient of OU3. These sources include the Pitt Street area (described in Section 2.3 of the RI Report), as well as undefined sources that are contributing CVOCs to the groundwater extracted at the Park Avenue Wellfield. The presence of these off-site sources means that the bedrock aquifer cannot be restored to ARARs by taking action at OU3 alone.

5. Applicable or Relevant and Appropriate Requirements

A discussion of ARARs that would require a waiver and the rationale behind the request for an ARAR waiver is presented below.

5.1. Safe Drinking Water Act (40 CFR 141)

Maximum Contaminant Levels (MCLs) and non-zero Maximum Contaminant Level Goals (MCLGs) established under the Safe Drinking Water Act are chemical-specific requirements for the contaminants present in the groundwater in OU3. Since the bedrock aquifer at OU3 is used as a potable water supply, MCLs and MCLGs are considered to be applicable requirements. These requirements will not be met within the OU3 groundwater plume since restoration of the aquifer such that it meets ARARs, allowing use as a potable water supply without treatment at the wellhead, is not practicable due to:

1. The confirmed presence of DNAPL in the bedrock groundwater, and the recalcitrant DNAPL components contained in the rock matrix.

2. The long history of industrial use and associated releases at the Site.

3. The volume, depth, and nature of the contaminated media (i.e., rock matrix)

4. The complex bedrock geology, including bedding planes, an interconnected fracture network, and steeply dipping fractures.

5. The OU3 location, which has developed surface industrial, commercial, and residential surface features, as well as transportation infrastructure.

6. The presence of potential CVOC sources other than CDE OU3.

5.2. New Jersey Safe Drinking Water Quality Act (NJAC 7:10-16)

The New Jersey Safe Drinking Water Quality Act MCLs (NJAC 7:10-16), and the New Jersey Groundwater Quality Criteria (GQCs) (NJAC 7:9-6) set the requirements for drinking water quality in the State of New Jersey. By rule, these requirements are as stringent, or more stringent, than those promulgated by the federal Safe Drinking Water Act. A waiver of the New Jersey Safe Drinking Water Quality Act MCLs and the New Jersey GQCs will also be required for the same reasons listed above.

6. Justification for Technical Impracticability

A Technical Impracticability Waiver of specific ARARs is appropriate for the OU3 groundwater due to the infeasibility of restoring groundwater within a reasonable timeframe at a reasonable cost. There are significant Site-specific factors that limit the ability of available remedial technologies to achieve groundwater ARARs at the Site.

6.1. Site Conditions

Groundwater flow in the Passaic Formation occurs primarily through the fracture network. The network is composed of bedding parallel to sub-parallel fractures with steeply dipping joint sets and is highly conductive and interconnected, allowing for the horizontal and vertical movement of groundwater. The highly conductive fracture network allows for the vertical and horizontal advection of groundwater and aqueous mass. Because the fracture network is so pervasive, it provides a relatively large surface area for the VOCs to sorb onto and then diffuse into the rock matrix. The pore volume of the rock matrix is nearly two orders of magnitude larger than the fracture network porosity, allowing it to hold the majority of the contaminant mass. Once the aqueous mass has diffused into the rock, it is left nearly immobile because of the low hydraulic conductivity of the rock matrix.

The aqueous mass migration has also been influenced by withdrawals over time at the adjacent well fields. These changes in conditions are likely to cause advective redistribution of the aqueous mass. In areas where the concentration of the aqueous mass in fractures is greater than that in the adjacent matrix pore water, diffusion into the rock is occurring and attenuating the leading edge of the aqueous mass. Furthermore, back diffusion out of the matrix (pore water) is occurring in areas where the concentration gradient between the rock matrix and the aqueous phase in fractures supports the process, which may contribute to ongoing groundwater contamination over a very long period of time (usually in multi-century timeframes). As a result, the contaminated aquifer cannot be restored such that it meets ARARs, allowing use as a potable water supply without treatment at the wellhead, in a reasonable timeframe and at a reasonable cost.

6.2. Technology Limitations

As discussed in herein, contaminant fate and transport modeling results indicate that treatment of bedrock limited to the area beneath the overburden source area (i.e., at MW-14S/D) would have negligible impact on the remainder of the downgradient plume and would not result in the achievement of ARARs since the bedrock matrix itself is the source of the ongoing exceedence of ARARs. Therefore, to be potentially capable of meeting ARARs, a remedial technology would have to be applied over the *entire* OU3 area where bedrock matrix contamination contributes to ongoing exceedences of ARARs.

This would encompass an area of more than 150 acres and depths of more than 250 feet below ground surface. The implementation of any of in-situ remediation technology over such an area is not practicable.

In addition, to be successful, a remedial technology would have to be capable of treating contamination in the both the rock matrix and the bedrock fractures. To do this, the technology must be capable of contacting impacted areas and maintaining that contact over a long enough time period to promote treatment in the rock matrix. Based on the review of currently available remedial technologies, there are no technologies capable of achieving these metrics in fractured bedrock in full-scale implementation.

6.3. Stability of Groundwater Conditions

As discussed above, matrix diffusion causes the leading edge of aqueous mass to be strongly attenuated relative to the mean groundwater velocity in the fracture network. This is due to the combined effects of diffusion-driven mass transfer from the fractures into the rock matrix, contaminant sorption and degradation, and hydrodynamic dispersion. The stabilization of the leading edge of aqueous CVOC mass is supported by field observations at several long term study areas, as well as the data collected from OU3 during the RI. These data indicate that little, if any, additional aqueous CVOC plume migration is anticipated to occur. Accordingly, a defined TI Zone that includes the OU3 contaminated groundwater can be established and maintained.

6.4. Overburden Source Removal

When restoration of groundwater to beneficial uses is not practicable, USEPA protocols require that potential sources of contamination be addressed to the extent practicable. As discussed herein, it has been demonstrated that it is technically impracticable to address the chlorinated ethenes present in the bedrock matrix and groundwater. The USEPA is currently completing remedial activities at OU2 that address the highest concentrations of CVOCs and PCBs in the overburden soil at the former CDE facility. In the September 2004 Record of Decision (ROD) for OU2, one of the Remedial Action Objectives requires the remedy to "Reduce or eliminate the migration of Site contaminants from soil and debris to the groundwater." The remedy included four key components:

- Relocation of the tenants at the Hamilton Industrial Park, demolition of the buildings and removal of the PCB-contaminated building debris for off-site disposal;
- Excavation, for off-site transportation and disposal, of the Capacitor Disposal Area (CDA), an area of debris located in the rear of the facility;
- Excavation of the Principal Threats posed by the site for on-site treatment using low- temperature thermal desorption (LTTD), or off-site disposal for material not amenable to LTTD treatment; and
- Capping of the residual soil contamination to prevent direct contact or off-site migration of contaminants left on site.

The OU2 remedy has been performed in phases. The USEPA began relocation of the tenants in 2006, and completed the last relocation in the spring of 2007. The building demolition phase was performed first, allowing access to underlying contaminated soil that needed to be excavated later. This work was completed in 2008. The CDA was addressed next, resulting in the removal of approximately 14,000 cubic yards of contaminated debris.

Starting in 2008 and concluding in early 2012, the USEPA - utilizing USACE contractors - has excavated approximately 159,000 cubic yards of soil and debris containing elevated levels of PCBs and/or CVOCs that were either treated on-site using LTTD or shipped for off-site disposal. Soil and debris was removed down to bedrock in many OU2 locations, effectively removing sources of contamination to groundwater.

The final restoration of OU2 is in progress and expected to be completed later in 2012. This includes grading the site so drainage is directed to the stormwater detention basin and installation of an asphalt cap over approximately 20 acres. This action of capping and collecting the surface water drainage will prevent surface water from migrating into the overburden and then into the bedrock groundwater. These actions further support risk reduction to the OU3 groundwater.

7. TI Zone

The USEPA TI Guidance states that at sites where restoration of groundwater to its most beneficial use is technically impracticable, the area over which the decision applies (referred to as the TI Zone) generally will include all portions of the contaminated groundwater that do not meet ARARs. ARARs are waived inside the TI Zone and other measures, such as pathway elimination and/or administrative controls, are used to prevent exposure to human health and the environment. Outside of the TI Zone, ARARs will still apply.

In accordance with the TI Guidance, a TI Zone has been developed that meets these criteria. The parameters for the TI Zone are presented below.

7.1. Horizontal Extent

The OU3 RI identified VOC groundwater contamination in shallow, intermediate, and deep portions of the fractured bedrock aquifer. As discussed herein, the Site and other sources contribute to the VOCs in the bedrock groundwater at concentrations greater than ARARs. The historic pumping of the deep bedrock municipal supply wells influenced the regional and local groundwater gradients and, as a result, the horizontal (and vertical) extent of the VOCs in the bedrock groundwater. Pumping at South Plainfield (1952-1969) and Spring Lake (1964-2003) shifted groundwater movement at the former CDE facility toward the South Plainfield and Spring Lake well fields. Today, the Park Avenue and Tingley Lane wellfields influence regional and local hydrogeology.

The aerial extent of the TI Zone is presented on Figure 7-1. Factors included in the establishment of the TI Zone horizontal boundaries are presented below.

1. Where data permitted, the TI Zone boundaries were set using the largest extent of the combined 1 ug/L TCE concentration isopleth from RI Figures 5-11, 5-14, and 5-17, for the shallow, intermediate, and deep depth intervals, respectively. These RI figures are included in Appendix B of this TIER.

2. Groundwater samples collected from monitoring well ERT-8, which is south of the former CDE facility, did not contain VOCs. This well defines the southern edge of groundwater contamination associated with the former CDE facility. Groundwater samples collected from monitoring wells ERT-5, ERT-6, and MW-18, which are located within the Pitt Street Well Contamination Area that is west/southwest of the former CDE facility, contained several CVOCs at concentrations that exceed ARARs. There are several lines of evidence (See RI Report Section 5.13.2) that suggest the former CDE facility is not the source of impacts in these wells; however, the results are not conclusive. Therefore, the groundwater impacts at ERT-5, ERT-6, and MW-18 have been included in the impacts from the former CDE facility and are included in the TI Zone.

3. Groundwater samples collected from monitoring well MW-22, which is northeast of the former CDE facility, contained increasing concentrations of VOCs with increased depth. Based on the results of the RI, the aqueous mass observed in this well is from the former CDE facility. Based on groundwater flow modeling, this well is most likely proximate to the eastern edge of potential groundwater contamination associated with the former CDE facility and is included in the TI Zone.

4. Groundwater samples collected from monitoring well MW-23, which was installed in an attempt to evaluate the northern extent of groundwater contamination, contained TCE at concentrations ranging from 3.8 to 120 micrograms per liter (μg/L). The highest TCE concentration was detected in the deepest sampling port at 454 feet bgs. These TCE concentrations, which increased with depth, were an order of magnitude less than the concentrations in MW-20, which is the next closest well to the former CDE facility. These data suggest that monitoring well MW-23 is near the northern boundary of the groundwater contamination, but that contaminant mass has moved to the north beyond MW-23, toward the Park Avenue wellfield.

The TI Zone encompasses all areas where site-related VOCs (specifically TCE) are present at concentrations greater than ARARs. As noted above, this includes the areas delineated by monitoring wells to the west of the former CDE facility that may have been impacted by the contamination emanating from the Pitt Street Well Contamination Area. The TI Zone also includes areas to the east that may be impacted by other sources.

7.2. Vertical Extent

As shown on Figure 7-2, the vertical extent of groundwater VOC contamination was not defined in the deepest monitoring port (454 feet bgs) in monitoring well MW-23, which is the most down gradient monitoring well location. Spring Lake Well No. 9, which is known to have contained VOCs associated with the former CDE facility at concentrations greater than ARARs, has a total depth of 505 feet bgs. Therefore, the vertical extent of the TI Zone is assumed to be 505 feet bgs as there are no deeper data points in the area.

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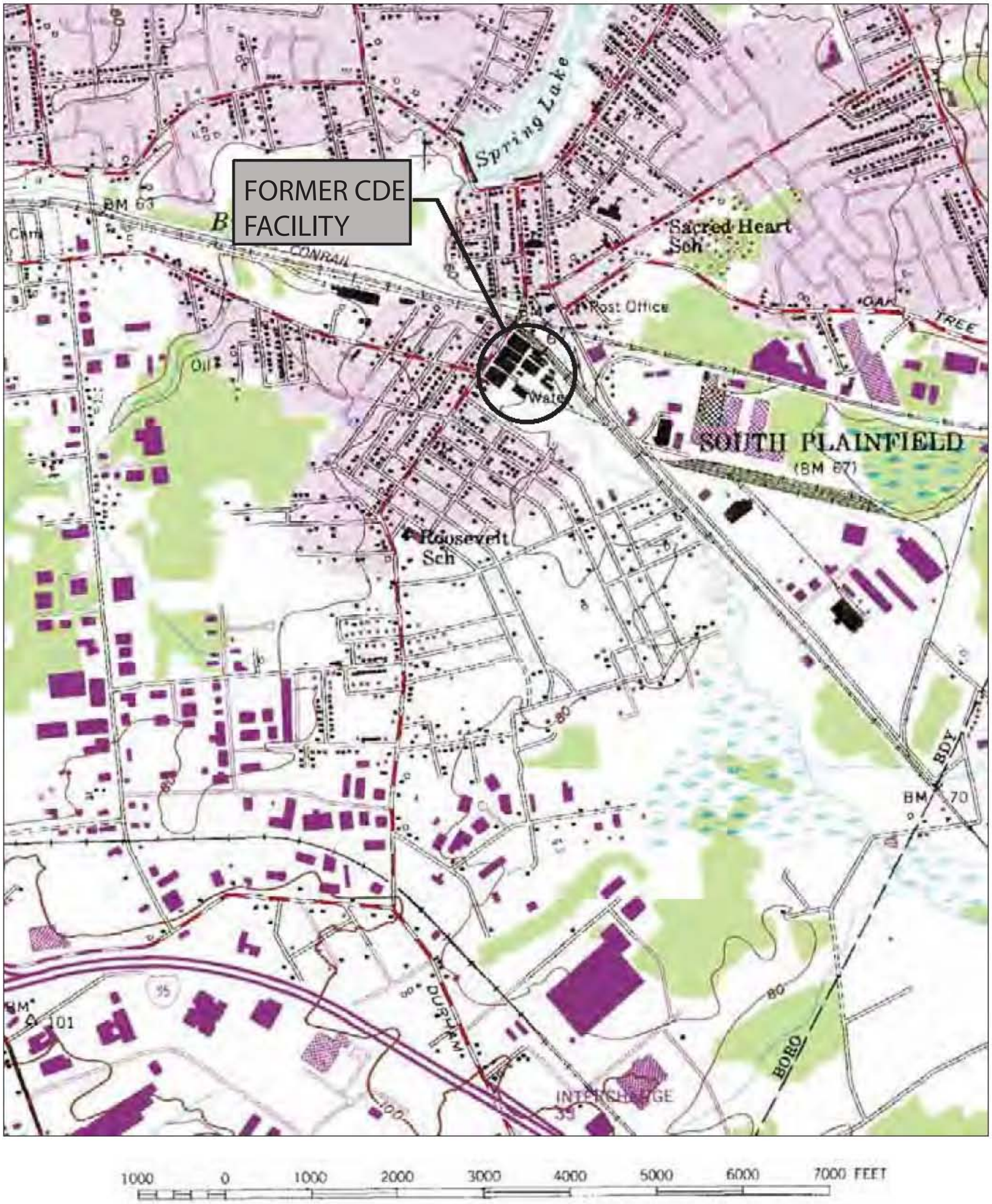
US Army Corps of Engineers

U. S. Army Corps of Engineers, Kansas City District –

CORNELL-DUBILIER ELECTRONICS SUPERFUND SITE
SOUTH PLAINFIELD, NEW JERSEY
TECHNICAL IMPRACTICABILITY EVALUATION REPORT
OPERABLE UNIT 3: GROUNDWATER

FIGURES

User: Monsanto Spec: PIRNIE STANDARD File: I:\ACAD\PROJ\4553033\CLIENT\BUFFALO\04-03-07\SITE-LOC-4553023-D.DWG Scale: 1:1 Date: 08/01/2007 Time: 15:30 Layout: FIGURE 1



SOURCE: U.S.G.S. TOPOGRAPHIC MAP,
7.5 MINUTE SERIES, PLAINFIELD, NEW JERSEY
QUADRANGLE, 1955, PHOTOREVISED 1981

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site South Plainfield, New Jersey	FORMER CDE FACILITY LOCATION MAP	FIGURE 1-1

LEGEND

Property Boundary

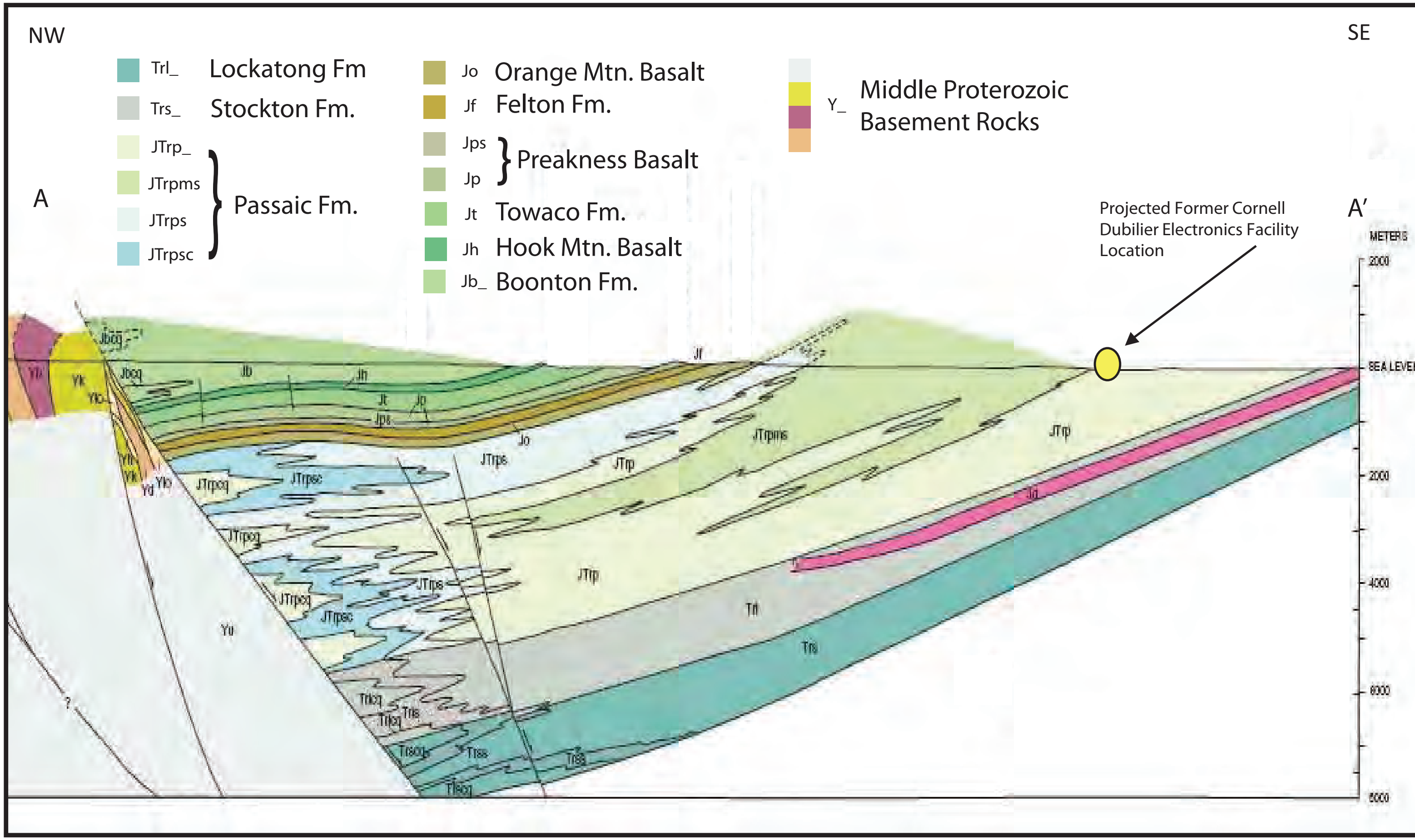
Bound Brook

0 250 500 1,000 Feet

Source: New Jersey Geographic Information Network
(NJ 2007 Ortho Imagery)

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site South Plainfield, NJ	2007 AERIAL PHOTOGRAPH	FIGURE 1-2

R2-0001500



Source: NJDEP GIS Database

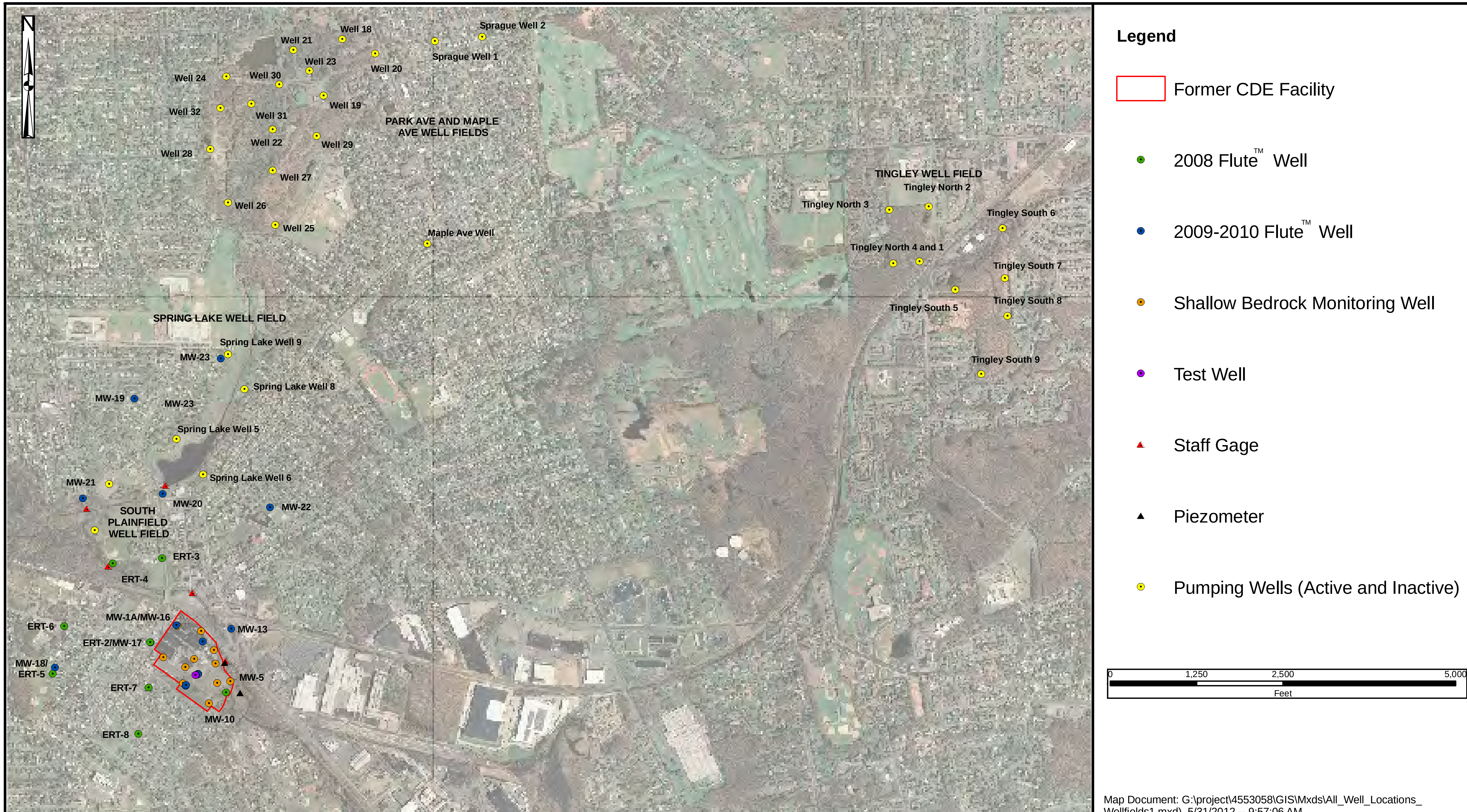
ARCADIS MALCOLM PIRNIE

Cornell-Dubilier Electronics Superfund Site
South Plainfield, New Jersey

Cross-Section of a Selected Portion of the Newark Basin

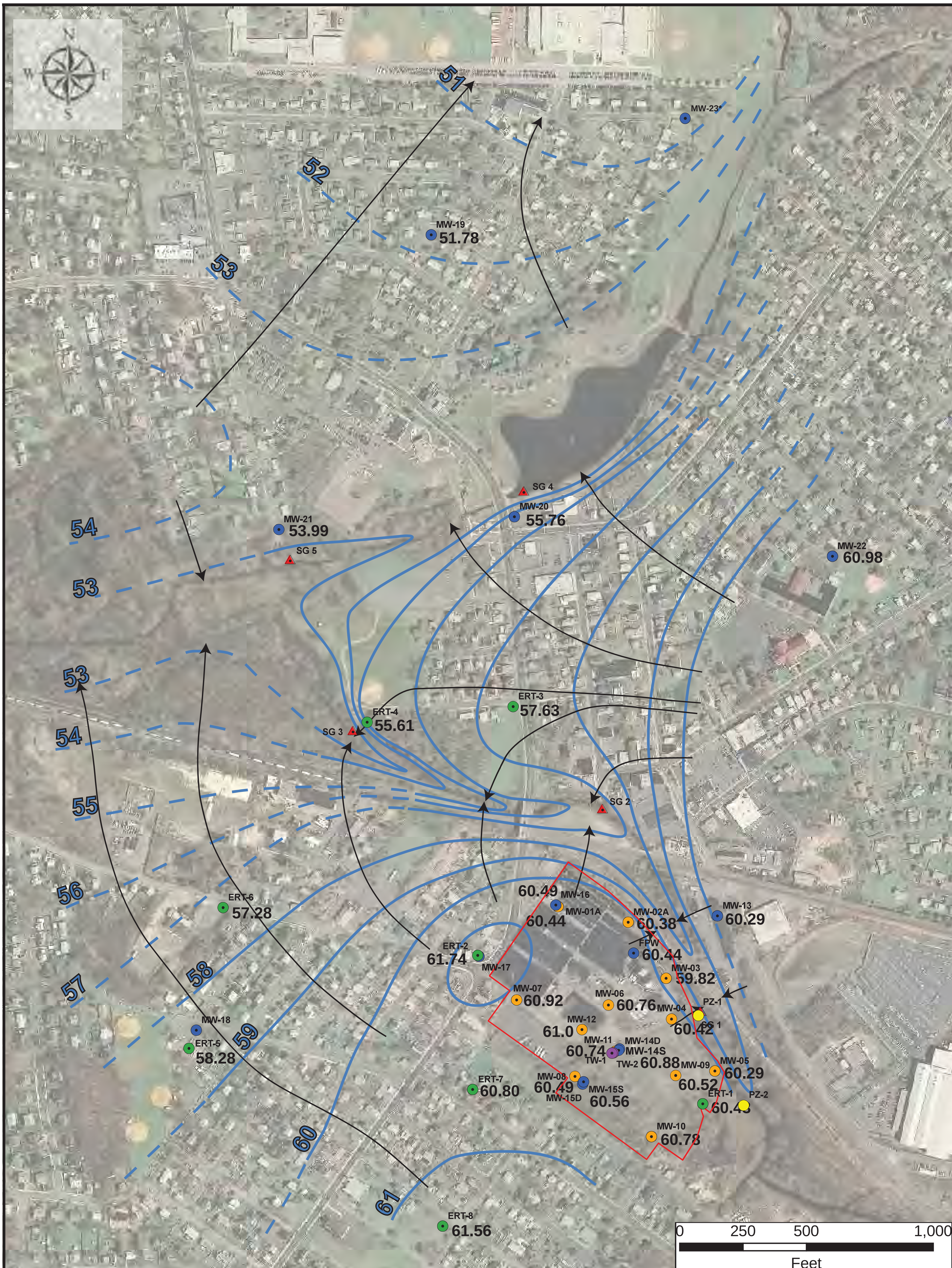
FIGURE 2-1

R2-0001501



ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site South Plainfield, New Jersey	Surrounding Wellfield Locations	FIGURE 2-2

R2-0001502



Legend

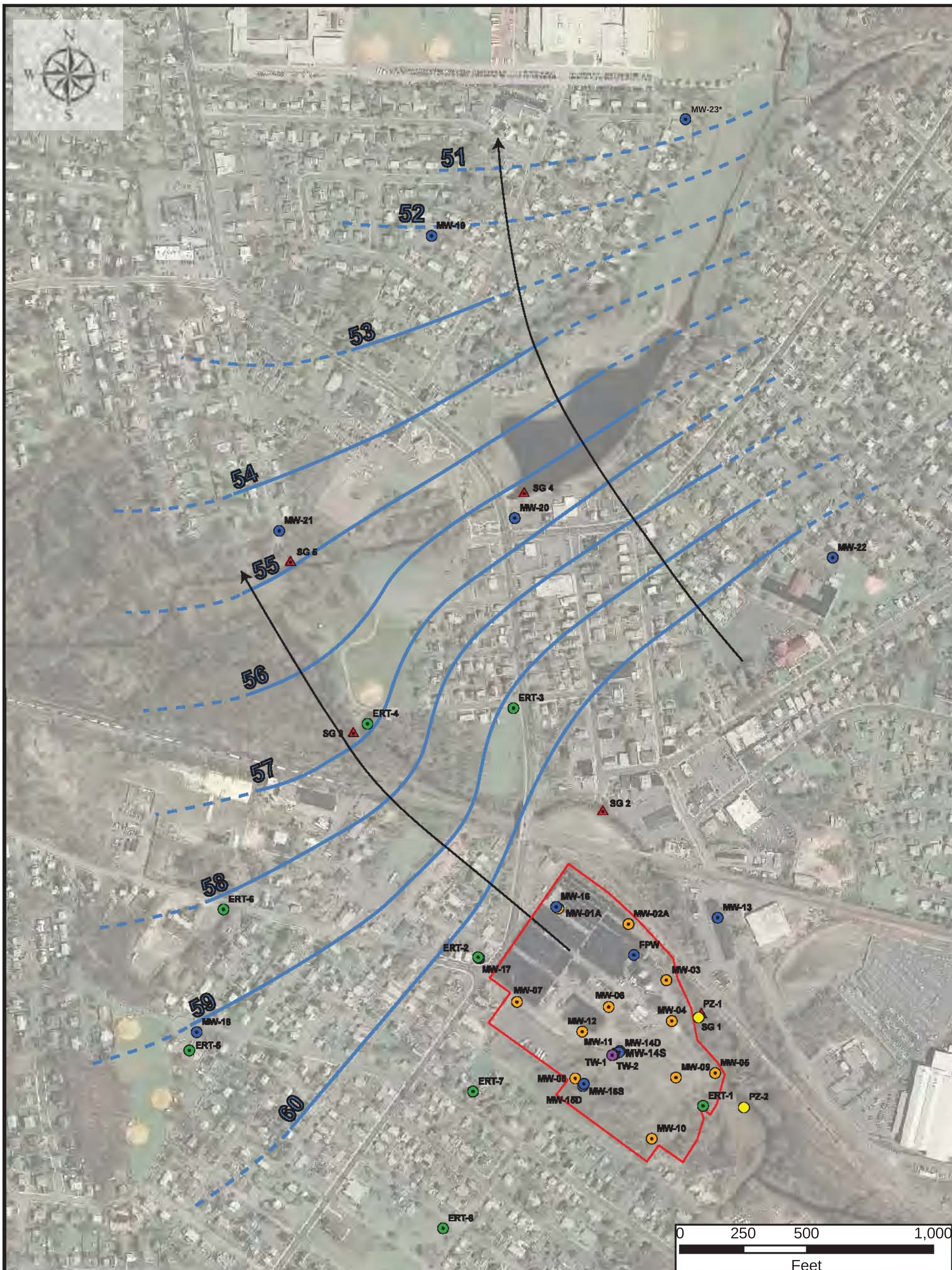
61.36 Groundwater Elevation (msl)

- Former CDE Facility
- Shallow Bedrock Monitoring Well
- 2008 Flute™ Well
- 2009 Flute™ Well
- Test Well
- Staff Gage
- Piezometer
- Direction of Groundwater Movement
- 61 Line of Equal Groundwater Elevation (ft msl) (dashed where inferred)

*Note: MW-23 installed and sampled in December 2010, March 2011

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site - OU3 South Plainfield, New Jersey	**Potentiometric Surface of Shallow (0' - 120' bgs) Water Bearing Zone July 9, 2010**	Figure 2-3

R2-0001503

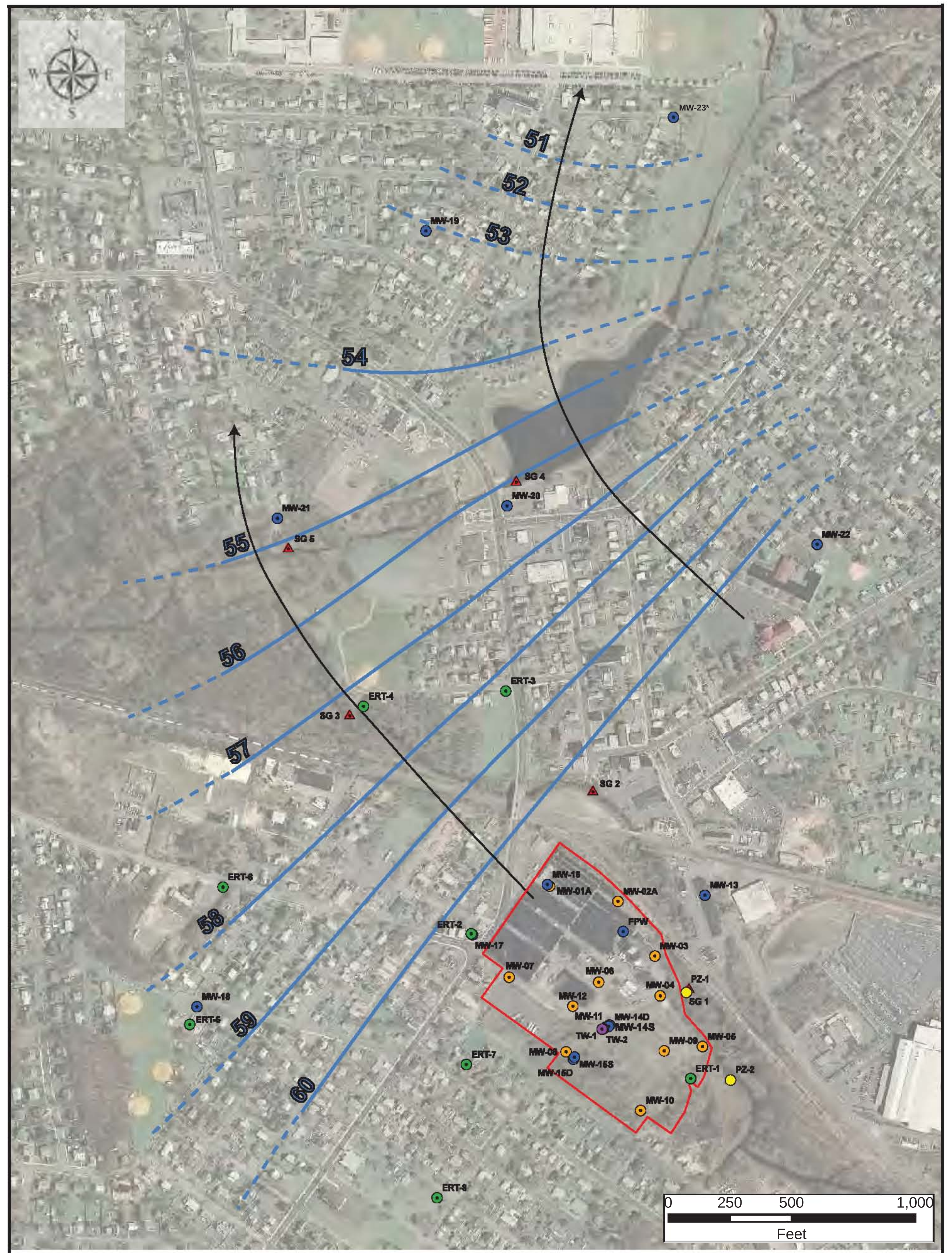


Legend

Former CDE Facility
Shallow Bedrock Monitoring Well
2008 Flute™ Well
2009 Flute™ Well
Test Well
Staff Gage
Piezometer
Direction of Groundwater Movement
61.36 Groundwater Elevation (msl)
61 Line of Equal Groundwater Elevation (ft msl) (dashed where inferred)

*Note: MW-23 installed and sampled in December 2010, March 2011

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site - OU3 South Plainfield, New Jersey	**Potentiometric Surface of Intermediate (120'-160' bgs) Water Bearing Zone July 9, 2010**	FIGURE 2-4



Legend

Former CDE Facility

Shallow Bedrock Monitoring Well

2008 Flute™ Well

2009 Flute™ Well

Test Well

Staff Gage

Piezometer

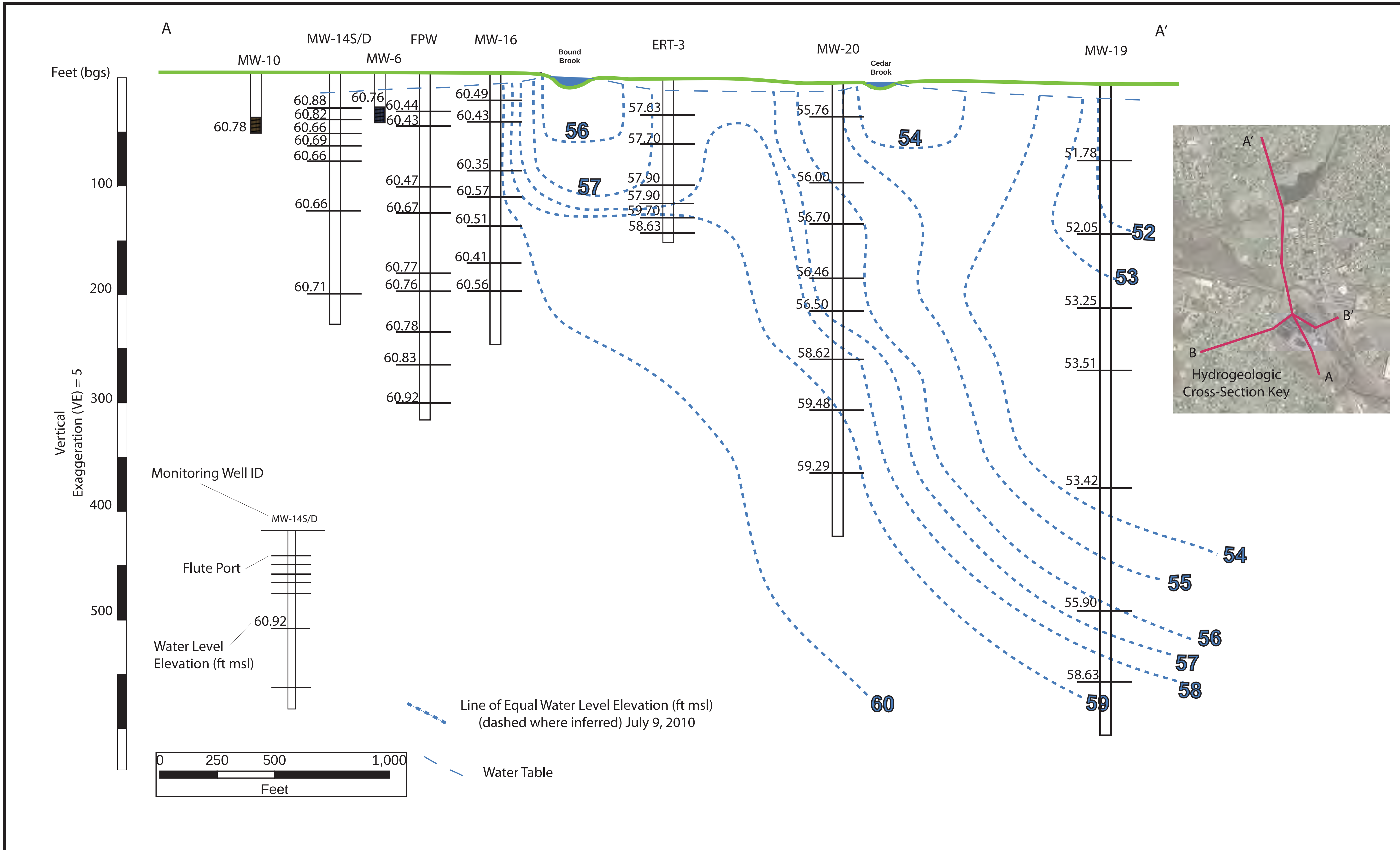
Direction of Groundwater Movement

61.36 Groundwater Elevation (msl)

61 Line of Equal Groundwater Elevation (ft msl) (dashed where inferred)

*Note: MW-23 installed and sampled in December 2010, March 2011

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site - OU3 South Plainfield, New Jersey	**Potentiometric Surface of Deep (200' - 240' bgs) Water Bearing Zone July 9, 2010**	FIGURE 2-5



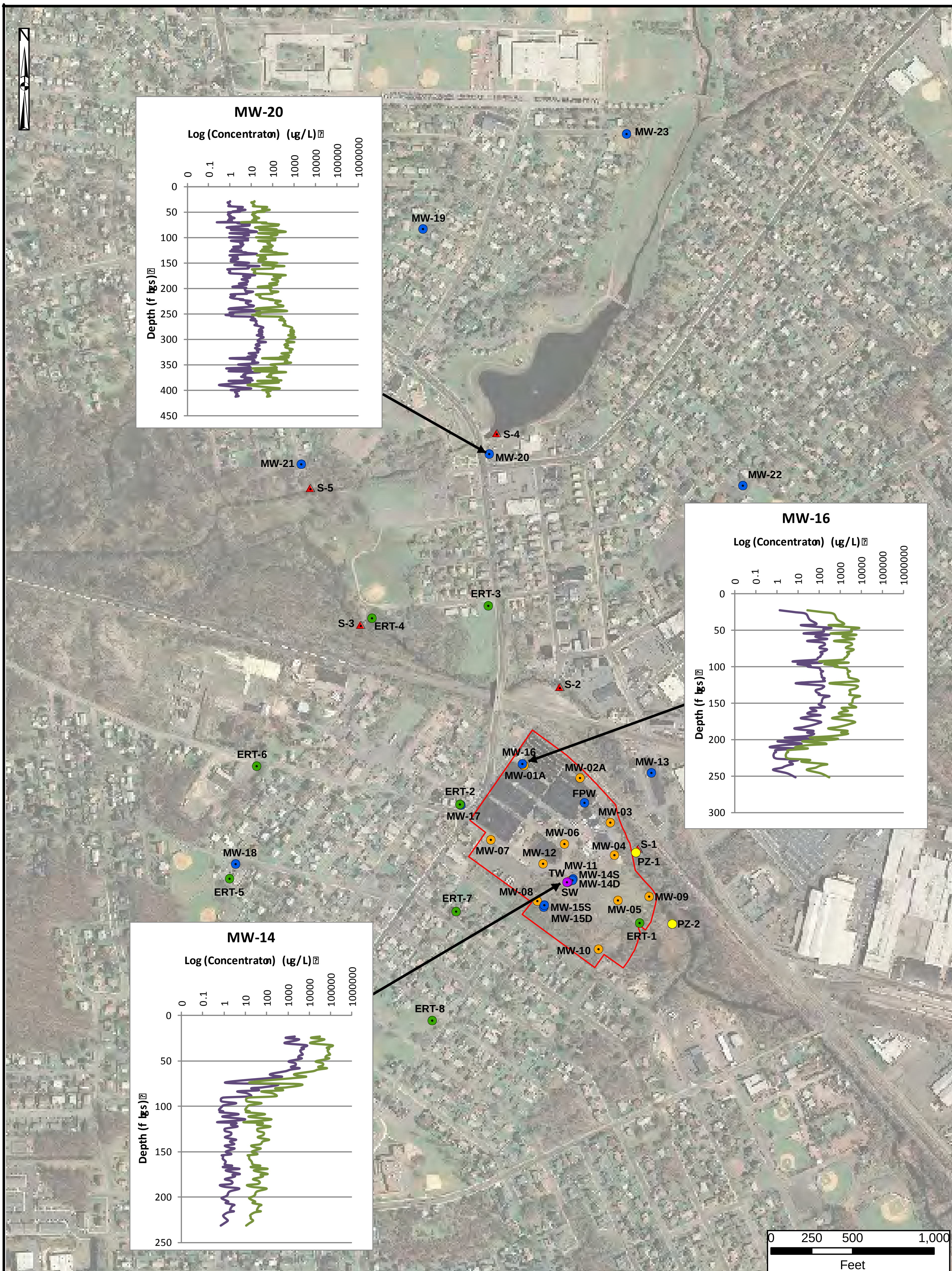
ARCADIS MALCOLM PIRNIE

CORNELL-DUBILIER ELECTRONICS SUPERFUND SITE - OU3 GROUNDWATER
South Plainfield, New Jersey

HYDROGEOLOGIC CROSS SECTION (N-S)
July 9, 2010

FIGURE 2-6

R2-0001506



LEGEND

Former CDE Facility	Shallow Bedrock Monitoring Well
2008 Flute™ Well	Test Well
2009-2010 Flute™ Well	Staff Gage
Piezometer	Estimated Pore Water TCE Concentration (ug/L)
	Rock Matrix Diffusion TCE Concentration (ug/kg)

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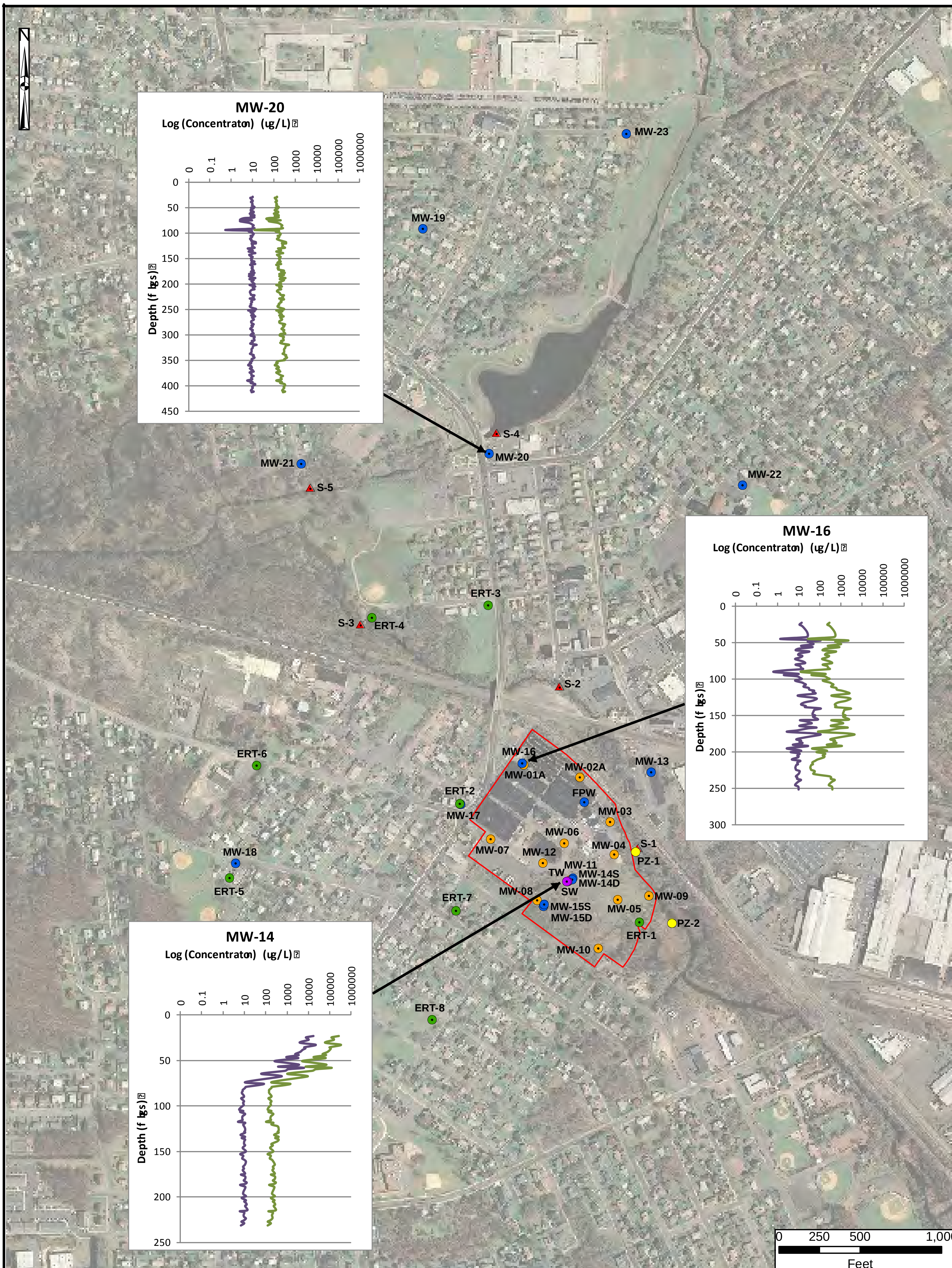
South Plainfield, New Jersey

Pore Water and Rock Matrix Trichloroethene Concentrations

FIGURE 3-1

Map Document: (G:\project\4553058\GIS\Mxds\NewTCE_PoreWaterandRockMatrix.mxd)
5/31/2012 -- 10:40:06 AM



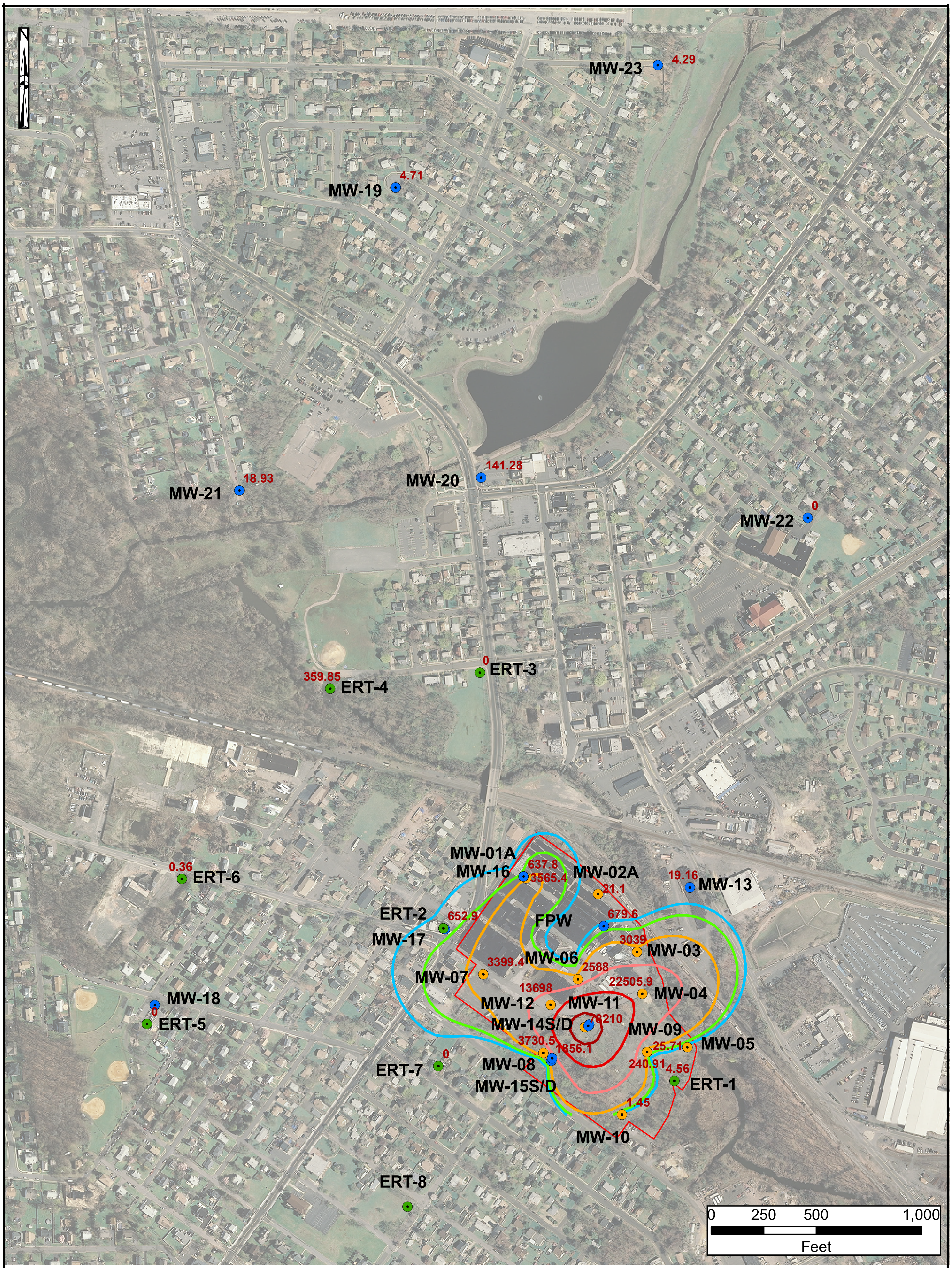
LEGEND

- Former CDE Facility
- 2008 Flute™ Well
- 2009-2010 Flute™ Well
- Shallow Bedrock Monitoring Well
- Test Well
- Staff Gage
- Piezometer
- Estimated Pore Water cDCE Concentration (ug/L)
- Rock Matrix Diffusion cDCE Concentration (ug/kg)

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site South Plainfield, New Jersey	Pore Water and Rock Matrix cis-1,2-Dichloroethene Concentrations	FIGURE 3-2

Map Document: (G:\project\4553058\GIS\Mxds\NewcisDCE_PoreWaterandRockMatrix.mxd)
5/31/2012 -- 9:40:06 AM

R2-0001508



Legend

- 2008 Flute™ Well
- 2009-2010 Flute™ Well
- Shallow Bedrock Monitoring Well
- Former CDE Facility
- 18.75 Total CVOC*, ug/L
- 500 ug/L
- 1000 ug/L
- 2500 ug/L
- 10000 ug/L
- 25000 ug/L
- 50000 ug/L

*All Data from March 2010 Sampling Event Except MW-23 (December 2010)

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Cornell-Dubilier Electronics Superfund Site
South Plainfield, New Jersey

Total CVOC Concentrations in Shallow Well Ports (0 to 120 ft bgs)

FIGURE 3-3

Map Document: (G:\project\4553058\GIS\Mxds\PlumeContours_Shallow.mxd)
5/31/2012 -- 9:57:06 AM

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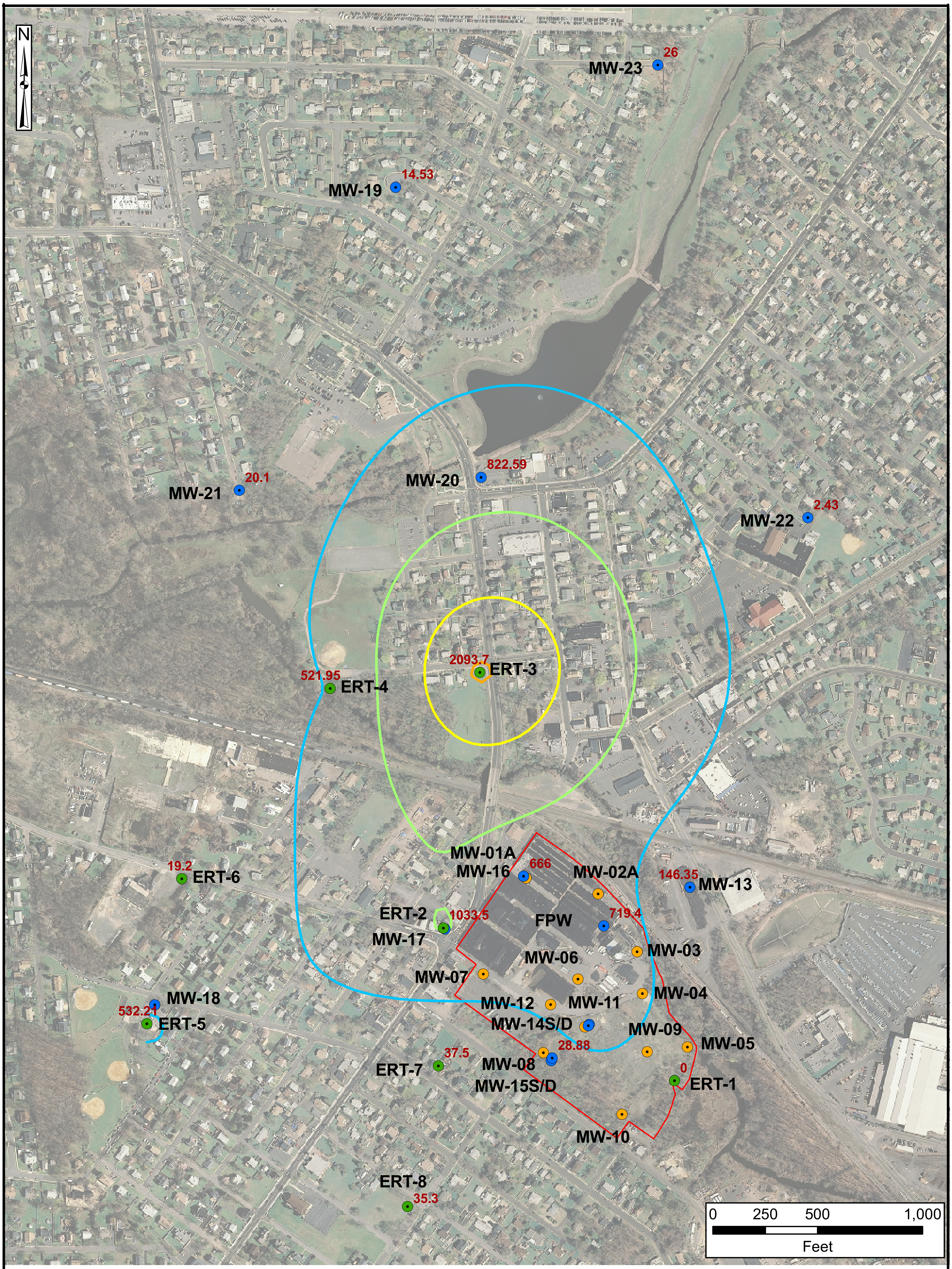


Legend

Former CDE Facility	2009-2010 Flute™ Well	**18.75** Total CVOC*, ug/L
2008 Flute™ Well	Shallow Bedrock Monitoring Well	

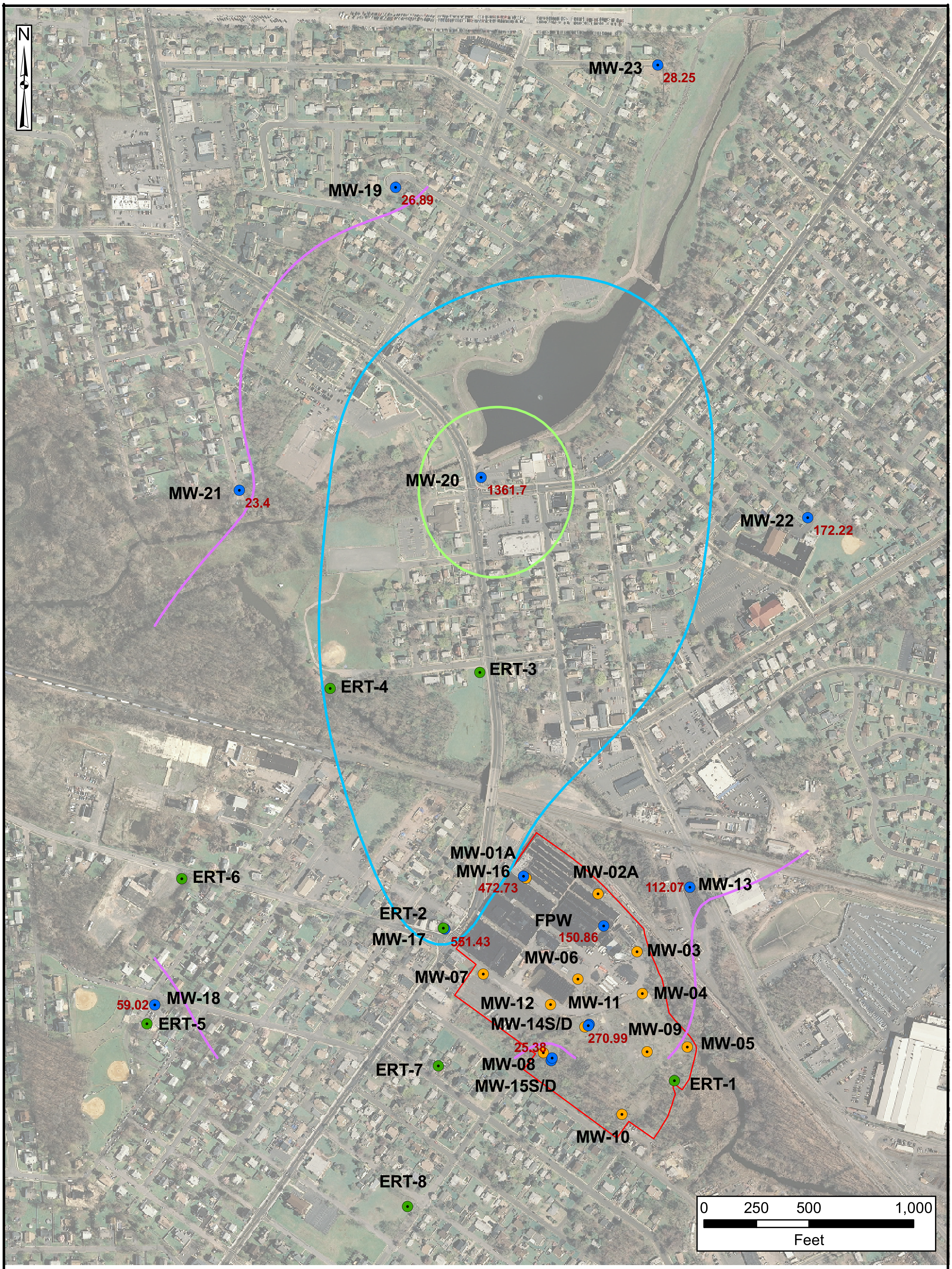
500 ug/L — 1000 ug/L — 1500 ug/L — 2000 ug/L

*All Data from March 2010 Sampling Event Except MW-23 (December 2010)

ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site South Plainfield, New Jersey	Total CVOC Concentrations in Intermediate Well Ports (120 - 160 ft bgs)	FIGURE 3-4

Map Document: (G:\project\4553058\GIS\Mxds\PlumeContours_Int.mxd)
5/31/2012 -- 9:57:06 AM

R2-0001510



*All Data from March 2010 Sampling Event Except MW-23 (December 2010)

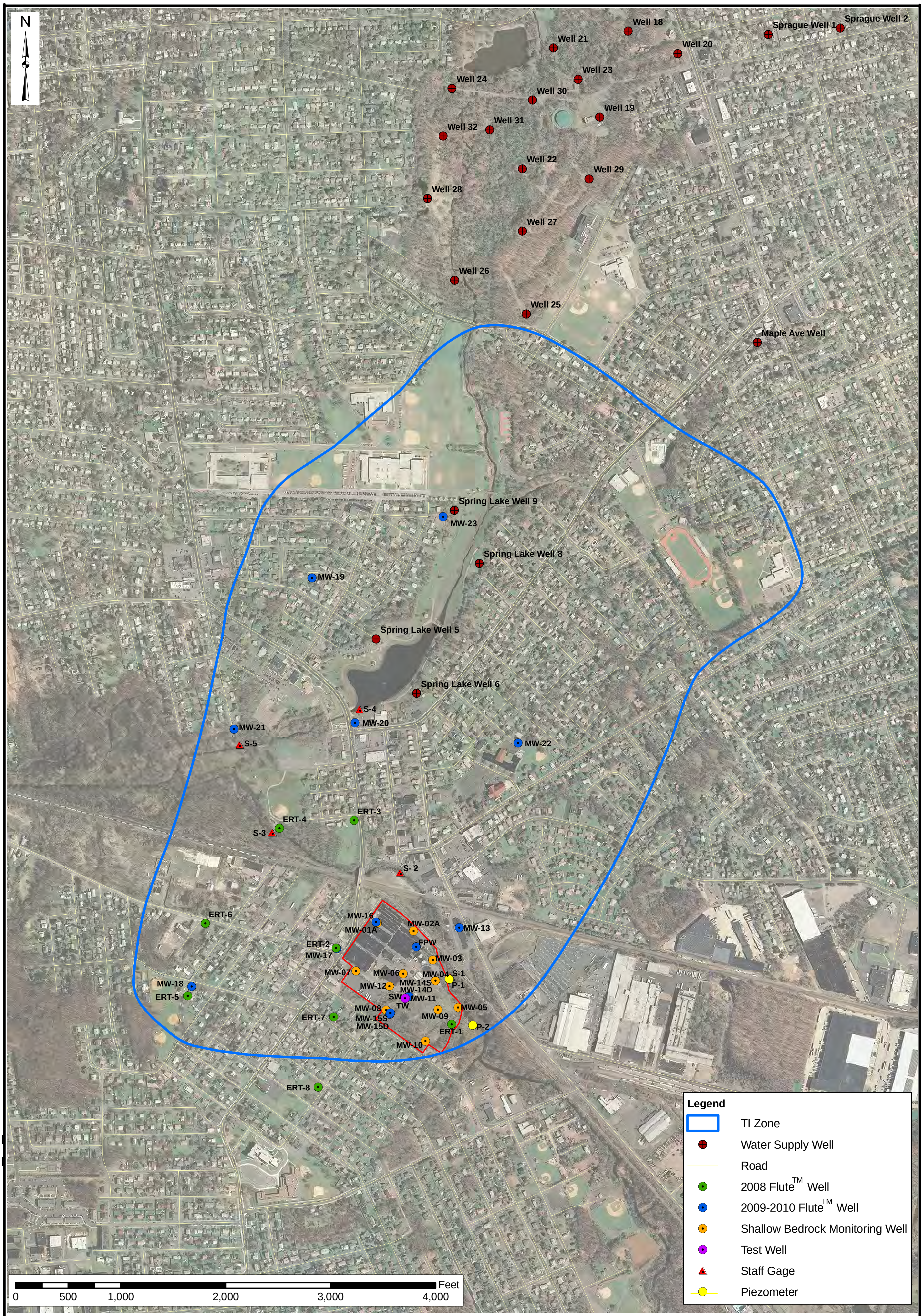
Legend

Former CDE Facility	2009-2010 Flute™ Well	18.75 Total CVOC*, ug/L
2008 Flute™ Well	Shallow Bedrock Monitoring Well	100 ug/L
500 ug/L	1000 ug/L	

ARCADIS MALCOLM PIRNIE Infrastructure · Water · Environment · Buildings	Cornell-Dubilier Electronics Superfund Site South Plainfield, New Jersey	Total CVOC Concentrations in Deep Well Ports (200 to 240 ft bgs)	FIGURE 3-5

Map Document: (G:\project\4553058\GIS\Mxds\PlumeContours_Deep.mxd)
5/29/2012 -- 9:57:06 AM

R2-0001511



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Cornell-Dubilier Electronics Superfund Site

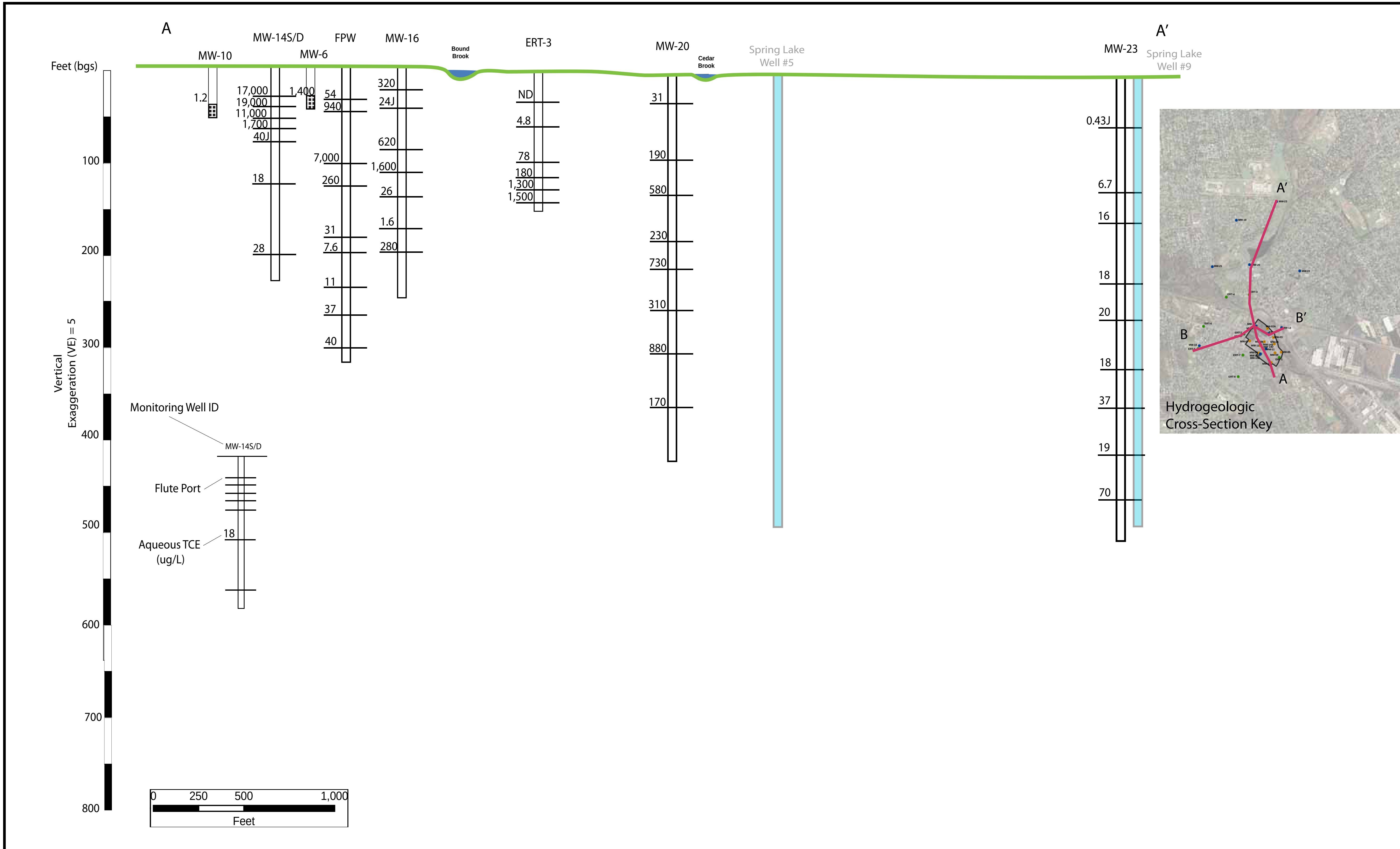
South Plainfield, New Jersey

OU3 TI ZONE

Figure 7-1

G:\GISMOD\06739004.0000\RevOU3_TI_Zone.mxd

R2-0001512



ARCADIS MALCOLM PIRNIE

CORNELL-DUBILIER ELECTRONICS SUPERFUND SITE - OU3 GROUNDWATER
South Plainfield, New Jersey

OU3 - TI ZONE CROSS SECTION with
AQUEOUS CONCENTRATION - TRICHLOROETHENE (TCE)

FIGURE 7-2

R2-0001513

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CORNELL-DUBILIER ELECTRONICS SUPERFUND SITE
SOUTH PLAINFIELD, NEW JERSEY
TECHNICAL IMPRACTICABILITY EVALUATION REPORT
OPERABLE UNIT 3: GROUNDWATER

APPENDIX A

Report on Discrete Fracture Network (DFN) Contaminant Transport Modeling

Cornell-Dubilier Electronics Superfund Site – OU3 Groundwater

Prepared by Steven Chapman, M.Sc.

Reviewed by Dr. Beth Parker and Dr. John Cherry

June 30, 2011

OVERVIEW

This report provides an overview of the methodology and results of discrete fracture network (DFN) simulations of contaminant fate and transport at the Cornell-Dubilier Electronics Superfund Site (the CDE site) in South Plainfield, New Jersey. This report is intended to be included as an appendix to the RI/FS reports. The technical memo submitted to EPA in February 2011 outlined the proposed modeling approach, where the bulk groundwater flow system is represented in a MODFLOW equivalent porous media (EPM) model followed by application of a discrete fracture network (DFN) model for assessing contaminant fate and transport in OU3 groundwater. The MODFLOW modeling report is provided separately as an attachment to the RI report. Data collected as part of the site investigations provided the necessary parameters for DFN simulations, including properties of the rock matrix (e.g. porosity, organic carbon content) and fracture network characteristics (e.g. fracture apertures and spacing). DFN simulations incorporate diffusion processes involving transfer of contaminant mass from fractures to the rock matrix, which has important implications for plume attenuation (e.g. Lipson et al., 2005) and remedial efficacy (e.g. Parker et al., 2010).

Included as an attachment to this report are a series of nine short articles on various aspects of the DFN approach for investigation of contaminated sites. Article 1 provides an overview of the DFN approach and the other articles describe various aspects of this approach, several of which have been applied during investigations at the CDE site including: use of rock core VOC analyses (Article 2); use of FLUTe liners for obtaining depth discrete measurements of permeability and for estimation of fracture apertures (Article 3); borehole geophysics (Article 5); and use of multilevel well systems for hydraulic head and groundwater sampling (Article 7). Article 8 provides an overview of the nature of chlorinated organic source zones and plumes in sedimentary rock, which is relevant to the CDE site conditions. Article 9 provides an overview of the DFN modeling approach which is the focus of this report.

EPM-DFN MODELING APPROACH

Pumping tests at the CDE site show that the groundwater flow system in the highly fractured bedrock can be reasonably simulated as an equivalent porous media (EPM). However, evaluation of contaminant fate and transport must consider effects of matrix diffusion on contaminant behavior in discretely fractured rock systems. While fractures provide the dominant pathways for groundwater flow (i.e. fracture porosity, which typically ranges from 10^{-3} to10^{-5}) the large rock matrix porosity (typically 2-20% in sedimentary rock such as sandstone, siltstone and shale) represents the bulk of the contaminant mass storage capacity. Thus diffusion of contaminants into the rock matrix in this dual porosity system, as well as sorption within the matrix and potentially contaminant degradation, is expected to have a strong influence on contaminant behavior and remedial efficacy. The attached Article 8 provides a more detailed overview of the nature of source zones and plumes in fractured sedimentary rock.

The modeling approach applied at the CDE site involved application of the MODFLOW EPM model to simulate the groundwater flow system to obtain overall bulk flow characteristics (i.e. hydraulic gradients, bulk hydraulic conductivity and groundwater fluxes) and then the discrete fracture network (DFN) model FRACTRAN was used to simulate contaminant fate and transport. Other data collected as part of the RI investigations (e.g. bulk hydraulic conductivity derived from FLUTe liner hydraulic conductivity profiling and pumping tests) also provide insight into the bulk groundwater flow system. Site investigations included application of field and laboratory testing to provide parameter inputs necessary for DFN simulations, including information on fracture apertures and rock matrix parameters. The attached Article 9 provides a more detailed overview of the EPM-DFN modeling approach.

The purpose of the DFN transport simulations is to represent groundwater flow and contaminant transport in fractured porous media incorporating relevant processes of rapid groundwater flow in fractures and contaminant diffusion into and out of the rock matrix. Other work has shown that matrix diffusion and degradation strongly affects contaminant transport in such dual porosity systems, with important implications for plume attenuation (e.g. Lipson et al., 2005) and remedial efficacy (e.g. Parker et al., 2010).

DFN simulations were conducted using the numerical model FRACTRAN, which was developed at the University of Waterloo based on Sudicky and McLaren (1992). FRACTRAN can be used to simulate steady state groundwater flow and transient contaminant transport in discretely fractured rock within a system of orthogonal fractures in 2-D. The model allows incorporation of fracture network geometry and relevant processes that will affect the transport of contaminants via interactions with the rock matrix (e.g. matrix diffusion, sorption, degradation) in discrete fracture networks in a much more realistic way compared to approaches that utilize dual-porosity methods. The National Research Council (NRC, 1996) provides an overview of the various simulation approaches. Deterministic simulations of contaminant transport in fractured rock are clearly not feasible given the complexity of fractured rock systems. However, FRACTRAN simulations can be used to represent site conditions in a 'stylistic' sense and are bounded by real data and incorporate site-specific inputs to the extent possible. Comparisons with field data can be performed, for example, with hydraulic head profiles in multilevel wells and contaminant distribution from multilevel wells and rock core sampling, to examine whether simulations reasonably represent field conditions. Overall this approach in coupling the two models, MODFLOW for the flow system and FRACTRAN for contaminant transport, is expected to provide a much improved understanding of controls on contaminant behavior. It is also a useful tool for assessing remedial options and efficacy.

DFN MODEL SETUP AND PARAMETERS

The FRACTRAN simulations were applied in vertical cross-section representing conditions along the approximate centerline of the plume flowpath. Fracture network characteristics (e.g. fracture network geometry – spacing, lengths and apertures) were constrained by field data to the extent feasible. Fracture spacing was based on core observations and geophysics, and fracture apertures derived from hydraulic testing. Groundwater flow rates and hydraulic conditions in the FRACTRAN DFN simulations were also constrained by the MODFLOW EPM simulations (see the MODFLOW report in the RI Appendix). While the FRACTRAN model is limited to 2-D domains with orthogonal fracture networks, fractures can have variable lengths, apertures and spacing and therefore can incorporate some of the complexity of real fractured rock systems. Following is an overview of parameter measurements on site samples and hydraulic testing data used for designing the DFN simulations. More detailed presentation and analysis of this data is provided in the RI report.

Rock matrix parameters

The hydrogeologic setting of the site is dominated by a dual porosity aquifer comprised of fractured mudstone (Figure 1) with appreciable matrix porosity. Table 1 provides a summary of laboratory physical property measurements performed on forty-one intact samples of rock core retained during the drilling at MW-14, MW-16 and MW-20. Figure 2 shows histograms for selected parameters. Rock matrix porosity (ϕ_m) ranged from 6 to 17% with an average of 10%. Total organic carbon (TOC) content ranged from 0.0025 to 0.033% with an average of 0.010% (excluding one outlier at 0.200% from MW-16). Assuming linear equilibrium sorption with partitioning dominated by organic carbon and using the well-known relation:

$$R = 1 + \frac{\rho_b}{\phi} K_{oc} f_{oc} \qquad [1]$$

the estimated retardation factor (R) for TCE is 1.3 applying average values for matrix porosity (ϕ_m), bulk density (ρ_b) and organic carbon content (f_{oc}), which is taken as the TOC value and using a literature organic carbon partitioning coefficient (K_{oc}) of 126 mL/g (from Table A1 of Pankow and Cherry, 1996). Applying values that would provide the highest possible R value (i.e. highest ρ_b and f_{oc} and lowest ϕ) the TCE retardation factor would be 2.8. For the FRACTRAN

DFN simulations an R factor for TCE of 1.5 was assumed. Measurements of rock matrix hydraulic conductivity (K_m) and tortuosity (τ) were not performed on CDE samples, and values of $K_m=1x10^{-8}$ m/sec and $\tau=0.10$ were assumed, which are consistent with literature values.

Fracture Network Characteristics and Bulk Hydraulic Conductivity

FLUTe liner hydraulic conductivity profiling was conducted at 22 corehole locations and results are summarized in Table 2. The attached Article 3 and Keller et al. (2011) provide more details on conducting and interpreting these types of tests, and more details on CDE site tests are provided in the RI report. The tests provided a range in overall bulk hydraulic conductivity (K_b) of nearly two orders of magnitude from $6.5x10^{-7}$ to $3.3x10^{-5}$ m/sec (0.2 to 9.2 ft/day) with an average of $7.7x10^{-6}$ m/sec (2.2 ft/day) and a histogram of the results is provided in Figure 3a. The average K_b is nearly 3 orders of magnitude higher than the estimated rock matrix hydraulic conductivity applied in DFN simulations, which is expected since the bulk hydraulic conductivity of such a highly fractured bedrock system is governed nearly entirely by the interconnected fracture network.

Hydraulic apertures were estimated for discrete features from the dataset by assuming that any sequential transmissivity values in the FLUTe datasets over short vertical intervals were attributed to a single fracture, and transmissivity of this zone was lumped to provide an assumed transmissivity value for the fracture (T_f), using the cubic law:

$$T_f = K_f(2b) = \frac{\rho g (2b)^3}{12\mu} \qquad [2]$$

where 2b is the hydraulic fracture aperture, K_f is fracture hydraulic conductivity, and μ is water viscosity. It should be noted that use of FLUTe liner profiling data for estimation of apertures for discrete fractures is a recent development, and rigorous review of methods for assessing such datasets to estimate apertures and comparison with more established methods such as packer testing have not yet been conducted. Figure 4 shows a histogram of estimated fracture apertures for all core holes tested, showing an overall range from <5 to 1300 microns with a geometric mean of 74 microns. Bulk fracture porosity (ϕ_f) was then estimated for each of the coreholes tested by summing all of the estimated apertures and then dividing by the length of borehole tested. These estimates indicate the bulk fracture porosity falls within a relatively narrow range from 1.2×10^{-4} to 5.2×10^{-4} with an average of 2.4×10^{-4} (see histogram in Figure 3b).

The bulk average linear groundwater velocity in the fracture network ($\bar{v}_f$) can be estimated using a modified version of Darcy's Law:

$$\bar{v}_f = \frac{K_b\, i}{\phi_f} \qquad [3]$$

where K_b is the bulk hydraulic conductivity, i is the hydraulic gradient and ϕ_f is the bulk fracture porosity. This calculation assumes that all flow occurs in the interconnected fracture network, and does not take into account lack of flow in dead-end fractures, flow in the rock matrix, and tortuosity of actual flow paths, and therefore only provides a rough estimate of average flow velocity through the fracture network. Applying the average K_b and ϕ_f values from the FLUTe liner testing and assuming an average hydraulic gradient of 0.3% (see potentiometric surface maps in the RI report and MODFLOW modeling report for more details on hydraulic gradients) provides an average linear groundwater velocity of 8.3 m/day, suggesting rapid groundwater flow rates in fractures. Therefore, in the absence of mass transfer via diffusion to the matrix and other attenuation processes, the plume would have been expected to travel long distances off-site reaching receptors (e.g. pumping wells or surface water) within relatively short periods of time after releases occurred.

Figure 5 shows a more detailed workup of data from one of the cored locations (MW-16). The first column shows fractures observed in cores (classified as 'horizontal', 'high angle' and 'broken zones'), fractures observed via acoustic televiewer (ATV) (classified as 'open' or 'less open') and fractures identified via the FLUTe liner testing. The interpreted FLUTe liner transmissivity profile for MW-16 is shown in the right hand column in Figure 5. At this location, the FLUTe test data apertures were interpreted in two ways, first using the methodology described above where any sequential transmissivity values were attributed to a single fracture, and then using a modified method assuming any transmissivity values falling within a 0.5 ft interval, and including any sequential T values falling outside this interval, were combined assuming a single fracture. The latter method is more conservative in that it yields fewer interpreted fractures and higher apertures (see fracture frequency comparison between the two methods in Figure 5). However, for the MW-16 dataset the difference in average apertures between these two methods was not that significant, with geometric mean apertures of about 52 and 60 microns for the first and second methods, respectively. The remaining FLUTe liner test datasets were interpreted using the first method only. Figure 5 also shows a comparison of fracture frequency estimated over 10 ft intervals based on core, ATV and FLUTe liner tests. In general it is expected that core data will overestimate in-situ fracture frequency since many of the core breaks observed may be mechanical breaks caused by drilling and extraction of cores from the core barrel, and ATV will generally underestimate fracture frequency due to inability of this technique to image smaller scale features (generally 2 mm resolution with fractures to 0.1 mm). Also, neither core nor ATV provides insight on whether groundwater flow occurs in the identified fractures (i.e. whether they are open with connectivity or closed).

As expected, at MW-16 fracture frequency via cores was generally higher in all intervals compared to the fractures identified with ATV. Fracture frequency derived from the FLUTe liner test data was also lower than that from core observations, particularly for the second method of lumping transmissivity values. Ideally the FLUTe liner test data would identify all permeable features, and it would be expected that core observations would overestimate frequency of transmissive fractures as discussed above. However, as discussed earlier, use of FLUTe liner test data to identify individual features is a recent development. The resolution and ability to identify individual fractures is affected by several factors including: 1) presence of high permeability zones which affects ability to resolve lower permeability zones, 2) time intervals used for logging liner descent, 3) 'noise' in the datasets due to operational variables, 4) borehole conditions (e.g. enlargements) which affect results, 5) assumptions in assigning transmissivity to individual features and transmissivity estimation methods, and 6) complexity of fractured rock systems. Therefore use of this data to assess individual features should be considered approximate and applied with caution.

Bulk groundwater flow conditions for use in the FRACTRAN DFN simulations were constrained based on the calibrated MODFLOW EPM flow model. It is reasonable to assume that an EPM model can provide bulk flow parameters (i.e. hydraulic gradients, bulk hydraulic conductivity and Darcy Flux along the plume flowpath) for conditions of dense, well-interconnected fracture networks. The attached Article 9 provides more details on the combined application of EPM models for flow and DFN models for contaminant transport. Figure 6 shows some of the MODFLOW results, including a plan view of the simulated potentiometric surface and position of the cross-section along the plume flowpath from the source area (Figure 6a) and flow pathlines in plan view (Figure 6b) and along the cross-section (Figure 6c). More details on this modeling are provided in the MODFLOW groundwater modeling report in the RI Appendices. The FRACTRAN DFN simulations cannot capture all of the complexity of the flow system simulated using MODFLOW, which includes surface water interactions with streams and a lake and historical pumping of various well fields. Thus, attempts were only made to represent average current conditions along the plume centerline, neglecting potential surface water interactions and historical pumping variations at the various well fields, such that simulations focus on longer-distance plume transport assuming current pumping at the Park Avenue well field dominates and would be the ultimate receptor. Based on the MODFLOW EPM flow simulation, groundwater flow conditions along an approximately 1400 m (4500 ft) long flowpath along particle traces released from the source (MW-14S/D) area extending to MW-23 can be represented by an average hydraulic gradient of 0.3% and bulk hydraulic conductivity ranging from $1.4\text{x}10^{-6}$ to $2.5\text{x}10^{-5}$ m/sec (0.4 to 7.0 ft/day) with a weighted average of $1.4\text{x}10^{-5}$ m/sec (4.0 ft/day). This average value is higher than the average estimated from the FLUTe liner tests of $7.7\text{x}10^{-6}$ m/sec (2.2 ft/day) (Table 2), but consistent with the pumping test results (see RI report). This is expected since the MODFLOW results represent larger scale values over the model domain.

Supporting information on bulk hydraulic conductivity of the bedrock from a transmissivity survey by the New Jersey Geologic Survey are summarized below (based on a 5-mile radius search of NJDEP Bureau of Water Allocation (BWA) records requested by Malcolm Pirnie):

1) Passaic Formation: mean=1193 ft^2/day, range=45-5362 ft^2/day, median=675 ft^2/day (N=19)
2) Brunswick Aquifer: mean=1091 ft^2/day, range=45-5362 ft^2/day, median=764 ft^2/day (N=27)

Assuming a range in well lengths from 300 to 500 ft provides a K_b range of 2.2 to 3.7 ft/day using the mean transmissivity value, and overall range from 0.1 to 18 ft/day applying the range in transmissivity values, which is generally consistent with values from the FLUTe liner testing and pumping test results.

Contaminant Conditions

The attached Article 8 provides an overview of a site conceptual model for chlorinated organic source zones and plumes in sedimentary rock. In this conceptual model, contaminant releases occurred as dense non-aqueous phase liquids (DNAPLs); however over a period of years to decades after releases occurred, the DNAPL mass becomes depleted due to dissolution in groundwater flowing in fractures and diffusion in the rock matrix, such that little to no DNAPL may remain at present time (Parker et al., 1994, 1997). Figure 7 shows estimated disappearance times for diffusion only using the average CDE site rock matrix parameters based on Parker et al., 1994. However, this analysis does not account for DNAPL dissolution in groundwater flowing through fractures, which can significantly decrease the time for DNAPL disappearance (e.g. Vanderkwaak and Sudicky, 1996). Thus, complete disappearance of DNAPL from fractures within a period of years to a few decades following releases is consistent with the rock matrix properties and groundwater flow rates at this site based on the range of fracture apertures estimated from FLUTe liner hydraulic conductivity profiling (i.e. <10 to a few hundred microns).

Rock core profiles of estimated TCE and cis-DCE porewater concentrations are plotted in Figure 8 for coreholes MW-14, MW-16 and MW-20, which represent conditions within the suspected source zone (MW-14) and at approximately 250 m (MW-16) and 800 m (MW-20) downgradient. Porewater concentrations were estimated from the total mass concentrations using partitioning calculations based on the rock matrix parameters and sorption estimates (more details are provided in the Stone Environmental report in the RI report appendices). The profiles in Figure 8 also show groundwater concentrations from two sampling episodes from the FLUTe multilevel wells (described by Cherry et al., 2007) later installed in these core holes.

Evidence for complete or nearly complete DNAPL disappearance is supported by rock core data collected from the suspected source area (combined profile from MW-14S/D; Figure 8) showing all estimated porewater TCE concentrations are below the aqueous solubility limit (~1100 mg/L; from Pankow and Cherry, 1996) with a maximum TCE of about 150 mg/L at 33 ft bgs (~13% of solubility) and most values one or more orders of magnitude below solubility. Similarly groundwater concentrations in the FLUTe multilevel well at this location were well-below solubility, with a maximum TCE concentration of 72 mg/L in the shallowest port (30-35 ft bgs). Recent observations suggested presence of residual DNAPL in the overburden in the area of

MW-14S/D, which was apparently mobilized into MW-14D based on NAPL reactive liner testing (see RI report for more details). However the groundwater and rock core data both do not support ongoing substantial DNAPL presence in bedrock, except for minor amounts that may have been mobilized into bedrock from investigation activities at MW-14S/D. Efforts were subsequently made to remove residual DNAPL in the overburden in this area via excavations to the bedrock surface (see RI report). Based on the strong concentration declines with depth at MW-14 based on both in the rock core data and FLUTe multilevel well groundwater data, it appears DNAPL penetration into bedrock may have been limited to the upper bedrock zone (i.e. upper 40 ft or less of bedrock). This limited penetration may have been controlled both by high horizontal fracture frequency and also by limited DNAPL release volumes. The RI report provides more information on site history and what is known about historical releases at the site. Recent remedial activities at the site have focused on removal of contaminated overburden to top of bedrock in the MW-14 area, as discussed more in the RI report.

The FRACTRAN DFN simulations were conducted for TCE only assuming no degradation, although FRACTRAN can accommodate first-order decay. Data from the site suggest transformation of TCE to cis-DCE occurs, but it is unknown whether much further dechlorination occurs since groundwater data shows little VC presence. More details on contaminant conditions are provided in the RI report. Therefore when comparing the FRACTRAN simulation data with field concentration data, the field data were converted to equivalent total TCE concentrations, assuming all cis-DCE observed was produced from TCE transformation, using the relation:

$$[Total\ TCE] = [TCE] + 1.35\,[cis - DCE] \qquad [4]$$

which corrects for the difference in molecular weights.

The FRACTRAN simulation results are assessed via: 1) ‘stylistic’ comparisons with total equivalent TCE based on the rock core VOC results at MW-14, MW-16 and MW-20 (Figure 9a) along the approximate plume centerline, and comparison with maximum observed concentrations versus distance based on both the rock core data and groundwater sampling data (Figure 9b). Both of these datasets show apparently strong attenuation in equivalent TCE concentrations with distance from the site. Maximum rock core equivalent TCE declines by nearly 3 orders of magnitude (OM) over the 800 m (~2600 ft) distance between MW-14 and

MW-20. Similarly, maximum equivalent TCE in groundwater declines by about 3OM between the source (MW-14) area and furthest downgradient monitoring well MW-23 positioned about 1400 m (~4600 ft) downgradient (north), with maximum equivalent TCE of about 150 μg/L.

Included in Figure 9b are projections of the field concentration data using the maximum estimated equivalent porewater TCE data from rock core, and the maximum equivalent groundwater TCE concentrations from multilevel wells (using only the highest concentration data from multilevel wells positioned along the inferred plume flowpath), beyond the distance of the Park Avenue well field, which is located approximately 2200 m (~7200 ft) downgradient. This simple analysis suggests it is possible that TCE emanating from the CDE site has not reached the well field at concentrations exceeding the MCL, although such projections are very uncertain given the complexity of fractured rock systems. Further interpretation of plume extent is provided in the RI report. It is expected that strong plume attenuation will occur due to diffusive mass transfer from groundwater flowing in fractures to the rock matrix (e.g. see attached Article 8; Lipson et al., 2005). The results of the FRACTRAN DFN simulations tailored to site conditions that follow can be used to assess the reasonableness of such projections.

FRACTRAN DFN SIMULATIONS

FRACTRAN Model Setup and Flow System

In FRACTRAN, the model domain (Figure 10a) for CDE site simulations is a vertical cross-section 1000 m long and 150 m high. The fracture network was selected after attempting several realizations of randomly generated fracture networks and adjusting the key fracture network statistics including mean fracture aperture and variance (Figure 11), fracture density and fracture length ranges to provide an overall horizontal bulk hydraulic conductivity within a target range based on the field data (e.g. FLUTe liner test data and pump test data) and MODFLOW EPM flow model results. Average hydraulic gradients in the FRACTRAN simulation were set to 0.3% (horizontal) and 0.3% vertical (downward) using constant head boundaries applied on all four sides of the domain. The average horizontal hydraulic gradient applied is consistent with the field head distribution and MODFLOW EPM flow system results (e.g. see Figure 6a). As described earlier, the FRACTRAN simulations cannot capture the full flow system complexity in the MODFLOW simulations including surface water interactions and effects of pumping of various well fields. The vertical head component was set to match the apparent plume deepening with depth based on the rock core VOC results (Figure 8, Figure 9a).

Based on steady state flow simulation results (e.g. see head distribution in Figure 12 for the final fracture network and boundary conditions selected) the horizontal K_b of the fractured rock system can be estimated using:

$$Q = K_b \, i \, A \rightarrow K_b = \frac{Q}{i \, A} \qquad [5]$$

where Q is the total simulated horizontal flow obtained by averaging inflow and outflow at the LHS and RHS of the model domain, respectively, or crossing the mid-point plane at X=500 m, and using the average horizontal hydraulic gradient (i) of 0.3% and cross-sectional area for flow (A) which is the 150 m domain height multiplied by unit thickness. Following flow simulations for several realizations, the selected fracture network (Figure 10a) has an overall horizontal K_b of 5.7×10^{-6} m/sec (1.6 ft/day), which is about 25% lower than the average determined from FLUTe liner testing (7.7×10^{-6} m/sec) and a factor of 2.5 lower than the mean from the calibrated MODFLOW EPM model (1.4×10^{-5} m/sec). The overall bulk fracture porosity (ϕ_f) of this fracture network is 1.5×10^{-4} (horizontal fractures~1.2×10^{-4}, vertical fractures~0.3×10^{-4}), which is lower than the estimates from the FLUTe liner test data (average of $2.4x10^{-4}$). This is expected based on the lower fracture density in the FRACTRAN generated fracture network. Justification for using a lower K_b for the model compared to field conditions is provided below.

While the FRACTRAN network has a relatively high fracture density, it is lower than the actual site fracture frequency (e.g. see Figure 5). Figure 10b shows example profiles of fracture positions and apertures along two vertical sections (X=250 m and 800 m), indicating an average fracture frequency of about 0.85 fractures per m (0.26 fractures per ft). Apertures in the FRACTRAN network (Figure 10a) are log-normally distributed with a geometric mean of 120 microns (Figure 11), which is higher than the mean from the FLUTe liner tests of 74 microns (Figure 4). The higher aperture applied in generation of the FRACTRAN network offsets the lower fracture density to increase the overall K_b. Incorporation of a higher fracture density was not feasible with the current version of FRACTRAN due to the higher grid discretization requirements. Together, the use of lower fracture density and higher mean apertures in the FRACTRAN simulations is expected to cause more rapid plume transport, and therefore a target K_b at the lower end of field estimates from FLUTe liner testing and pumping tests and applied in the MODFLOW EPM model was used to offset this effect. Applying the horizontal K_b, i and horizontal ϕ_f of the FRACTRAN network to estimate an average linear groundwater velocity in the fracture network using Equation 3 provides a value of about 12.3 m/day, which is larger than estimate based on the FLUTe data (8.3 m/day). The FRACTRAN network (Figure 10a) has lower fracture density and bulk fracture porosity compared to field estimates and higher mean apertures. The alternative method of lumping transmissivity values from the FLUTe liner test data, described above for MW-16, would provide a higher mean aperture and lower fracture frequency somewhat more in line with the FRACTRAN network. Despite the FRACTRAN network having lower K_b compared to the field estimates based on the FLUTe liner test data and pumping tests, the FRACTRAN network still has a larger average linear groundwater velocity due to the lower ϕ_f. This suggests potential for the FRACTRAN simulations to overestimate rates of plume transport compared to the field conditions.

FRACTRAN Contaminant Transport Simulations

In the FRACTRAN DFN simulations, the ‘source zone’ was positioned within the upper portion of the model domain (Figure 10a) consistent with the apparently limited DNAPL penetration. For simulation purposes it is assumed that DNAPL releases occurred about 50 years ago based on site history (described in the RI report), although earlier releases may also have occurred. The source input was assumed constant at aqueous solubility for a period of 20 years, representing a conservative estimate of the time for complete DNAPL disappearance, followed by a period of sustained input at 10% of solubility to present time, representing dissolution of remnant DNAPL in overburden (which presumably occurred in isolated zones) causing ongoing mass input into the upper bedrock zone.

Results of the FRACTRAN transport simulation are plotted in Figure 13 for times of 10, 25 and 50 years (with the latter assumed to represent near present time when rock core sampling was conducted to obtain the profiles shown in Figure 8(2009). FRACTRAN concentrations are plotted as relative concentrations assuming C_o=1.0 represents TCE aqueous solubility (~1100 mg/L). Profile results provided later are converted to aqueous concentrations by multiplying by this solubility value. The FRACTRAN results show a range in concentrations spanning 5 orders of magnitude, consistent with the difference between TCE solubility and its MCL (0.005 mg/L). As indicated by the DFN transport simulation results, contaminant migration in the fracture network is much slower than groundwater flow rates in fractures, due to attenuation processes including diffusion of mass from fractures to the rock matrix. However, by 10 years, the simulation results show contamination has already reached the model boundary at 1000 m, and by 50 years contamination occurs throughout the model domain. Ideally the FRACTRAN fracture network would be extended a sufficient distance to capture the full extent of plume transport and provide insight into the distance and rates of plume front migration (e.g. to assess whether contamination from the CDE site may have reached downgradient well fields). However, given the high fracture density and requirements for fine grid discretization to resolve diffusion processes in the matrix, it was necessary to limit the domain size so the code could be compiled with necessary array sizes. The grid for the current network contains nearly 4 million nodes (NX=3036, NZ=1280) and 600,000 line elements representing the fractures. There are plans to modify the code so larger arrays can be handled, which would allow the model domain to be extended and still incorporate a similar fracture density, but this is not yet available.

Figure 14 shows a comparison of the FRACTRAN simulated profiles at X=10, 250 and 800 m with the field rock core profiles at similar distances at MW-14, MW-16 and MW-20 (note different X-axis scales for simulated and field profiles at MW-16 and MW-20). FRACTRAN concentrations were converted from relative to aqueous concentrations by assuming C_o=1.0 represents TCE aqueous solubility. The field rock core profiles show estimated total equivalent TCE concentrations using partitioning calculations to estimate porewater TCE and cis-DCE concentrations (see RI report for more details) and then applying Equation 4. This comparison shows very good “stylistic” agreement between the FRACTRAN simulation results and field rock core profiles. For the source area (X=10 m versus MW-14) the concentration distributions and magnitudes are relatively similar, with the rock core profiles showing higher concentrations in a couple of samples adjacent to fractures. A short distance downgradient (X=250 m versus MW-16) the contaminant distributions are again quite similar stylistically, but FRACTRAN results overestimate the magnitude of the concentrations, with the concentration scale for the FRACTRAN spanning a range 5X greater than for the MW-16 profile. This is also the case further downgradient (X=800 m versus MW-20) where the concentration scale for FRACTRAN profile spans a range 10X greater than for the MW-20 profile. It should be noted that these results are for one realization of a random fracture network; profiles would be expected to vary for different realizations, but overall transport distances and migration rates should be fairly similar between realizations for fracture networks generated using the same fracture network statistics. Figure 15 shows a comparison of the maximum equivalent TCE concentrations versus distance for the field data (rock core VOC profiles and groundwater samples from multilevel wells along the approximate plume centerline) versus the FRACTRAN results at 50 years. The results are consistent in that they all show strong attenuation in maximum concentrations with distance downgradient from the source area. The FRACTRAN results are generally expected to be conservative (i.e. produce more rapid downgradient plume transport and higher downgradient concentrations compared to field conditions) based on a number of the FRACTRAN assumptions and factors not included in these DFN simulations, including:

1) FRACTRAN simulations are for a 2-D vertical cross-section domain, which:

a. assumes an infinitely wide source, which is not the case (see RI report for evidence of a fairly localized source in the MW-14 area);
b. neglects plume spreading in the transverse direction, which would have the effect of reducing concentrations downgradient due to increased mixing in the fracture network (i.e. between fractures having higher and lower concentrations) and increased attenuation due to matrix diffusion since the transverse spreading would increase the fracture-matrix contact area;

2) As discussed above the selected fracture network for the FRACTRAN simulations has a lower fracture frequency and higher mean apertures compared to field estimates (due to numerical limitations) and therefore these conditions would be expected to produce more rapid rates of plume migration. To some extent this effect was offset by using an overall lower K_b for the FRACTRAN fracture network, but the average groundwater velocity in the fracture network is still larger for the simulated scenario;
3) Simulations assume a constant source input for 20-year duration, in reality DNAPL may have disappeared from many fractures sooner than this causing reduced contaminant loading over time compared to the assumptions in the FRACTRAN simulation;
4) Simulations neglect degradation effects (but compare to equivalent total TCE based on TCE and cis-DCE concentrations) so any further degradation occurring is not reflected in the FRACTRAN simulations, and even very slow degradation rates can have strong attenuation effects when combined with matrix diffusion;
5) Simulations neglect flow system transience (e.g. due to variable pumping at different well fields over time) which is expected to have caused additional plume spreading and increased attenuation due to more contact area for matrix diffusion and more tortuous flow paths;
6) The maximum concentrations extracted from FRACTRAN simulations are actual ‘point’ concentrations, whereas field concentrations from monitoring wells or multilevel well ports are ‘blended’ values over larger vertical intervals.

Overall the FRACTRAN transport simulation results confirm the strong attenuation inferred based on the field data, showing matrix diffusion effects can account for such strong plume attenuation when combined with a finite source input. Given that the majority of contaminant mass now occurs in the rock matrix, mass discharge in downgradient portions of the plume may be relatively small. For example, based on the FRACTRAN results, the mass discharge in the downgradient portion of the plume at X=800 m at 50 years was assessed. Figure 16 shows profiles of fracture apertures, groundwater flow rates and concentration profiles at 50 years. Mass discharge within the plume across this plane was estimated by multiplying the nodal groundwater flow rates and concentrations and summing over the entire thickness. This provides an estimated TCE mass discharge at 50 years of about 0.3 kg/year per m width (since model domain is a vertical cross-section with unit thickness). With expectations of strong attenuation with distance, mass discharge would be significantly lower than this further downgradient, so that even if TCE contamination from the site has reached the Park Avenue well field, the resulting increase in concentrations may be very small when dilution effects from pumping are factored in.

Future Projection Scenarios

For future projections, two scenarios were assumed: (1) continued input at 10% of solubility, and (2) complete removal of the source input term. The latter scenario is consistent with the recent remedial efforts focused on contaminated overburden removal, assuming any remnant DNAPL in overburden is successfully removed and no longer contributes mass to the bedrock system. This could also represent a scenario where not all DNAPL is removed, but where a source zone hydraulic control system is put in place where any contaminant mass emanating from the source zone is captured and treated. Figure 17 shows simulated concentration contours for these two scenarios of continued source mass input versus complete removal of source mass input at times of 50, 100 and 150 years from present. The results show little impact of complete removal of source mass input on persistence of the downgradient plume, which is expected given that the majority of the contaminant mass exists in the rock matrix. Some minor improvements in internal plume water quality are evident, which are shown more clearly in the plume profile comparisons in Figure 18 at X=10 m, 250 m and 800 m. In these plots the “MNA” scenario assumes continued input while the “Source Removal” scenario assumes no continued input due either to complete removal of all DNAPL in overburden or hydraulic cutoff. While some minor improvements in groundwater quality internally within the plume are achieved from complete source removal or cutoff, the time to achieve such benefits are more than 100 years.

Actual source conditions at the CDE site are likely in between these two end points given recent efforts to remove contaminated overburden materials. While these FRACTRAN DFN simulations do not incorporate a sufficiently large domain to capture the full simulated plume extent, the expectation is that the rate of plume front migration would be very slow at present time due to effects of matrix diffusion. These simulations also suggest efforts to completely remove source inputs would have negligible impact on conditions nearer the plume front within any reasonable timeframe. Similar types of scenarios where a larger zone close to the former source is fully remediated (e.g. which could represent aggressive thermal treatment) are provided by Parker et al. (2010) which shows similar results of only minor improvements in downgradient water quality after extended periods of time and little to no effect on the plume front. Inclusion of slow degradation in simulations, if evidence suggests complete dechlorination were occurring in OU3 groundwater, would show more of an effect on the plume following source depletion or cutoff. However this does not seem to be justified based on site data collected to date, but could be examined in future simulations.

CONCLUSIONS

Overall, the FRACTRAN DFN simulations, tailored to site conditions to the extent feasible with the flow system constrained by the MODFLOW EPM results, show that matrix diffusion is expected to have strongly attenuated plume transport at the CDE site. This supports the field data showing strong declines in contaminant concentrations with distance from the site. Results indicate the majority of contaminant mass is now present in the rock matrix, such that mass discharge within the plume in fractures which carry the bulk of groundwater flow should be relatively low. The mass distribution also has significant implications for source zone and plume remediation efficacy. More interpretation of these results will be provided in the RI/FS reports.

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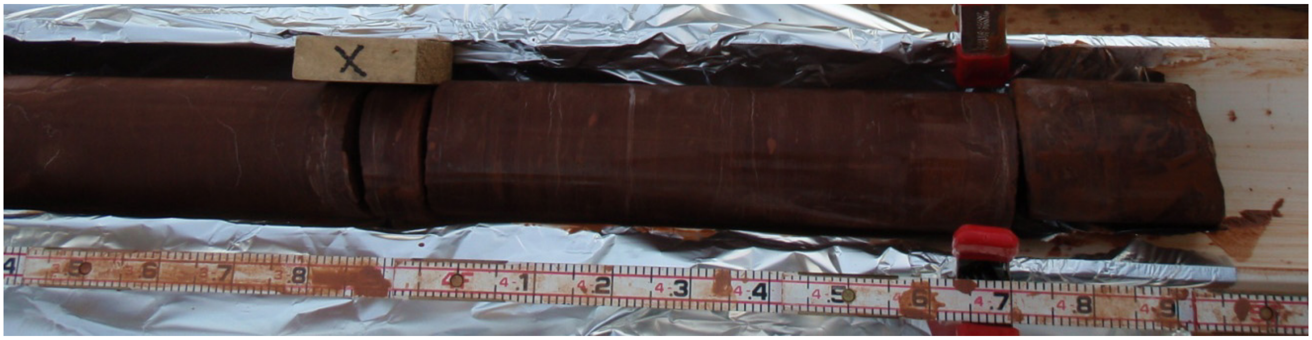
FIGURES

(a) MW-14 (Run#10, 64-69 ft bgs)



(b) MW-14 (Run#11, 73-78 ft bgs)

(c) MW-20 (Run#27, 158-163 ft bgs)

Figure 1.Representative core photos from the CDE site cores: (a) MW-14, Run# 10 (64-69 ft bgs), (b) MW-14, Run# 11, 73-78 ft), (c) MW-20, Run#27 (158-163 ft bgs).

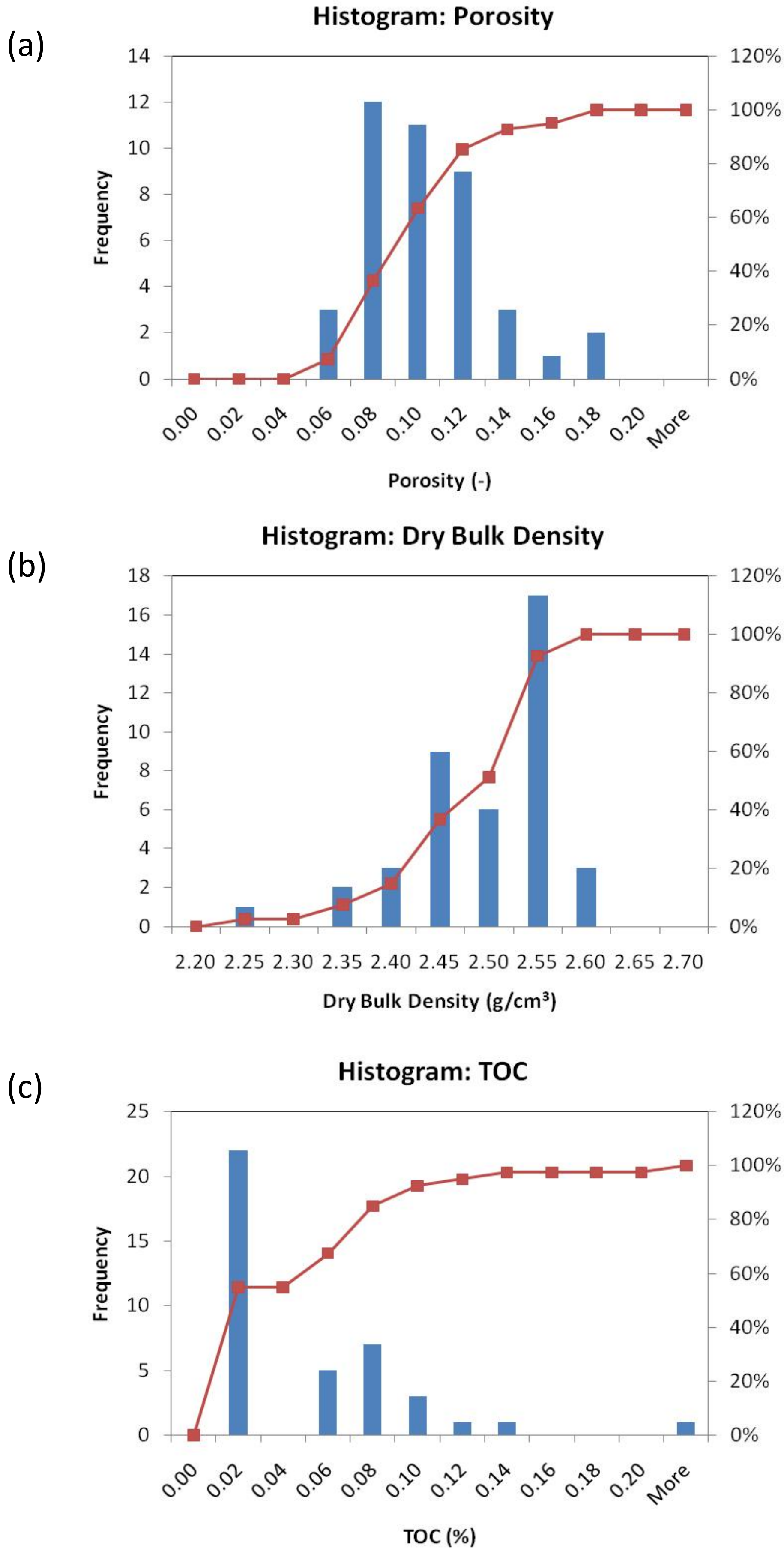
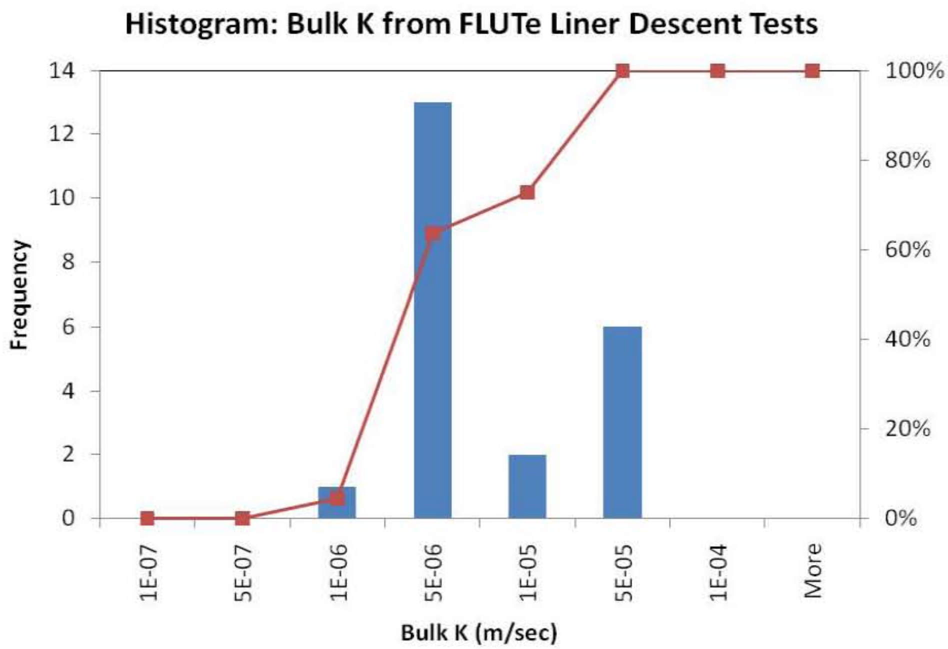


Figure 2.Histograms of (a) matrix porosity, (b) bulk density and (c) total organic carbon (TOC) based on measurements by Golder Associates on 41 samples from core holes MW-14, MW-16 and MW-20.

(a)



(b)

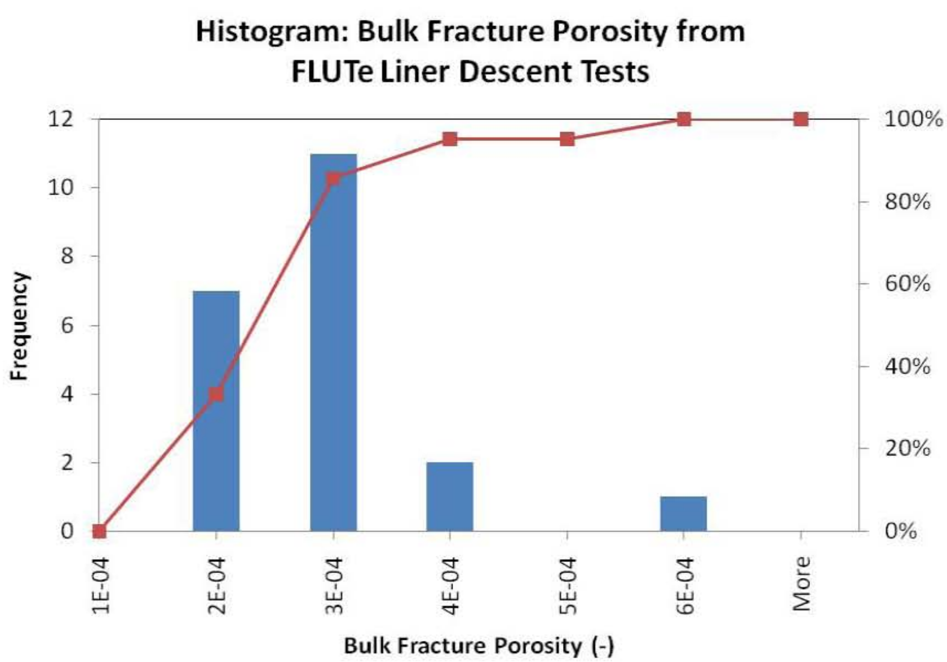


Figure 3. Histograms of (a) overall bulk hydraulic conductivity and (b) overall bulk fracture porosity estimated from the FLUTe liner descent tests conducted in 22 coreholes at the site.

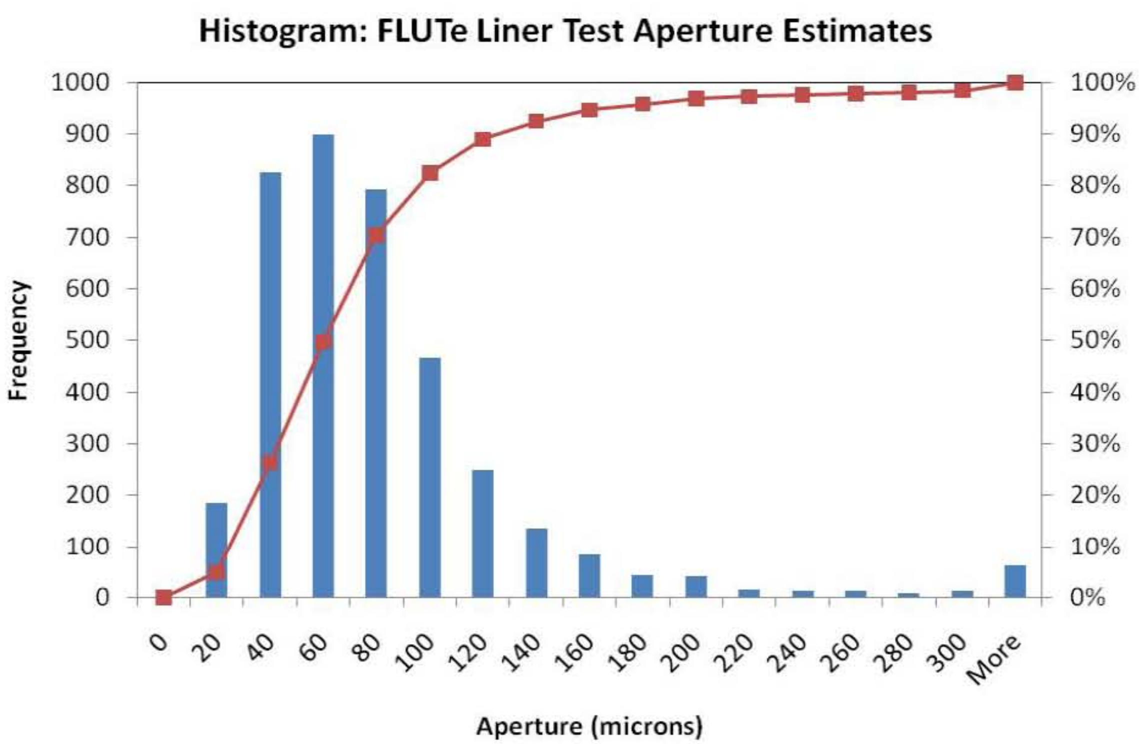


Figure 4. Histogram of individual fracture apertures estimated based on the FLUTe liner descent tests conducted in 22 coreholes at the site.

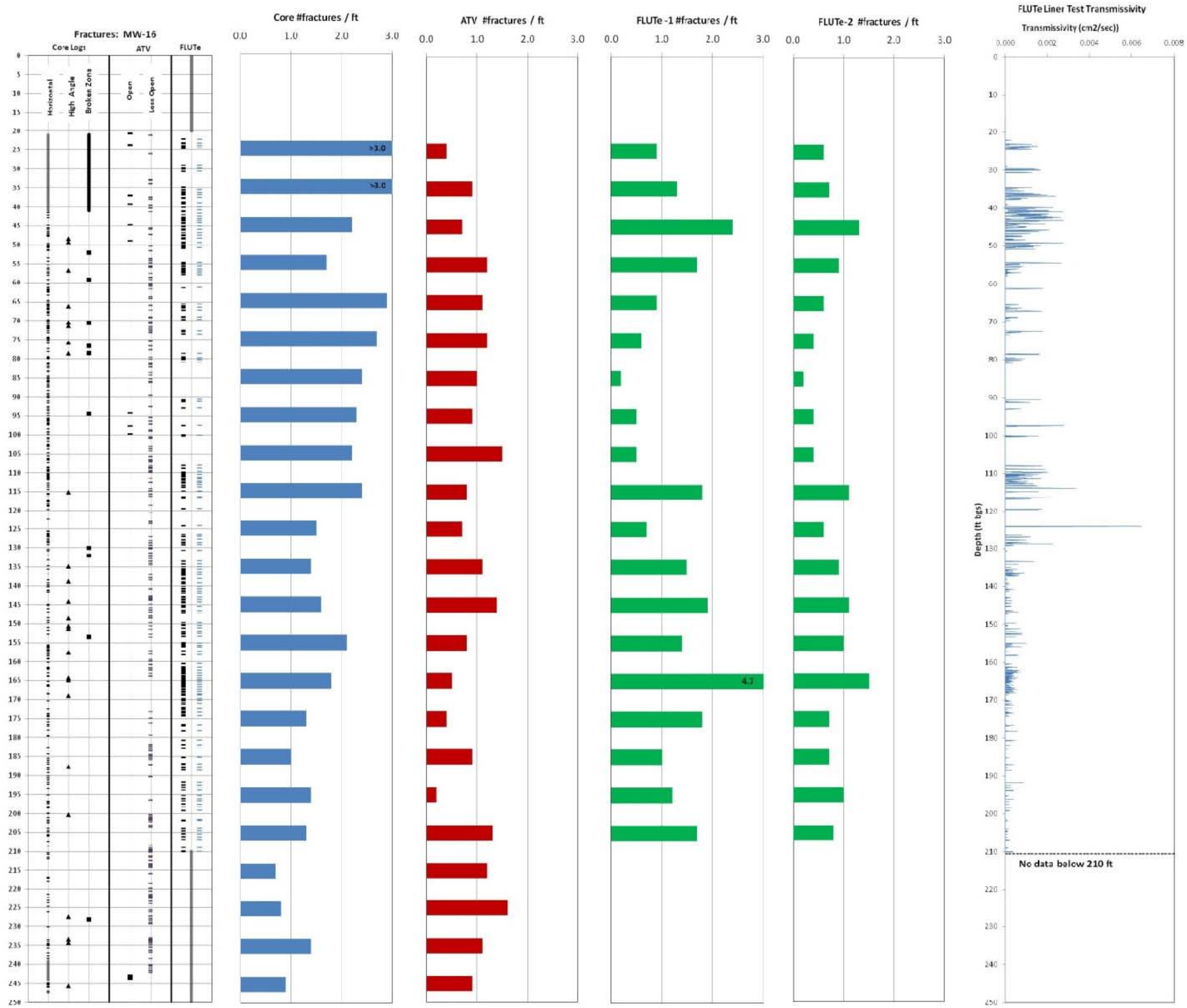
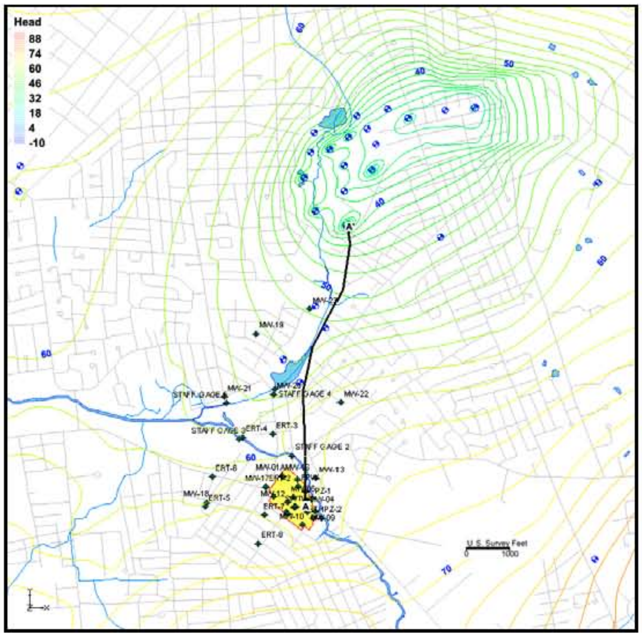
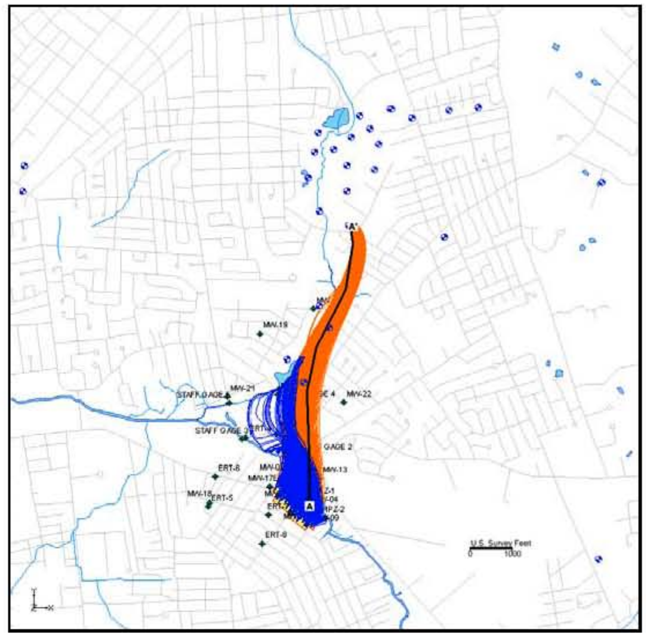


Figure 5. Example of fractures observed at MW-16 in cores and via acoustic televiewer and inferred from the FLUTe liner tests along with estimated fracture frequency (over 10 ft intervals) and transmissivity profile from the FLUTe liner testing.

(a)



(b)



(c)

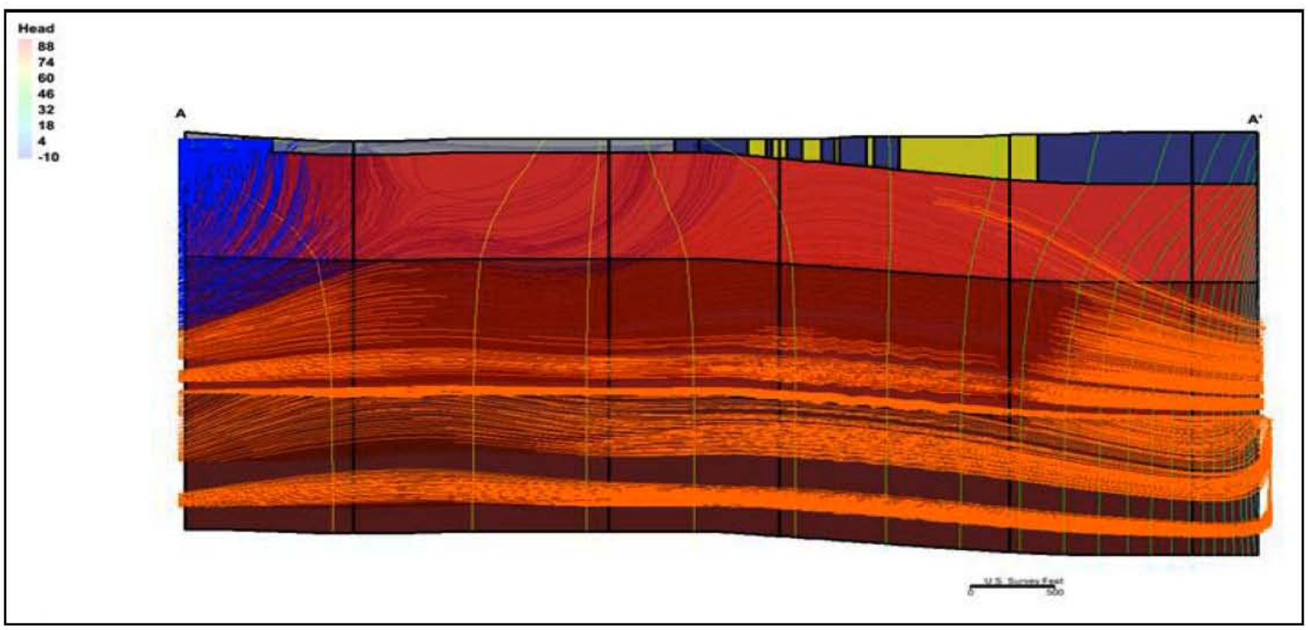


Figure 6. Selected results from the MODFLOW EPM simulations showing: (a) simulated potentiometric surface and position of the cross-section along the plume flowpath, and flow pathlines for particles released from the source area in (b) plan view and (c) along the cross-section.

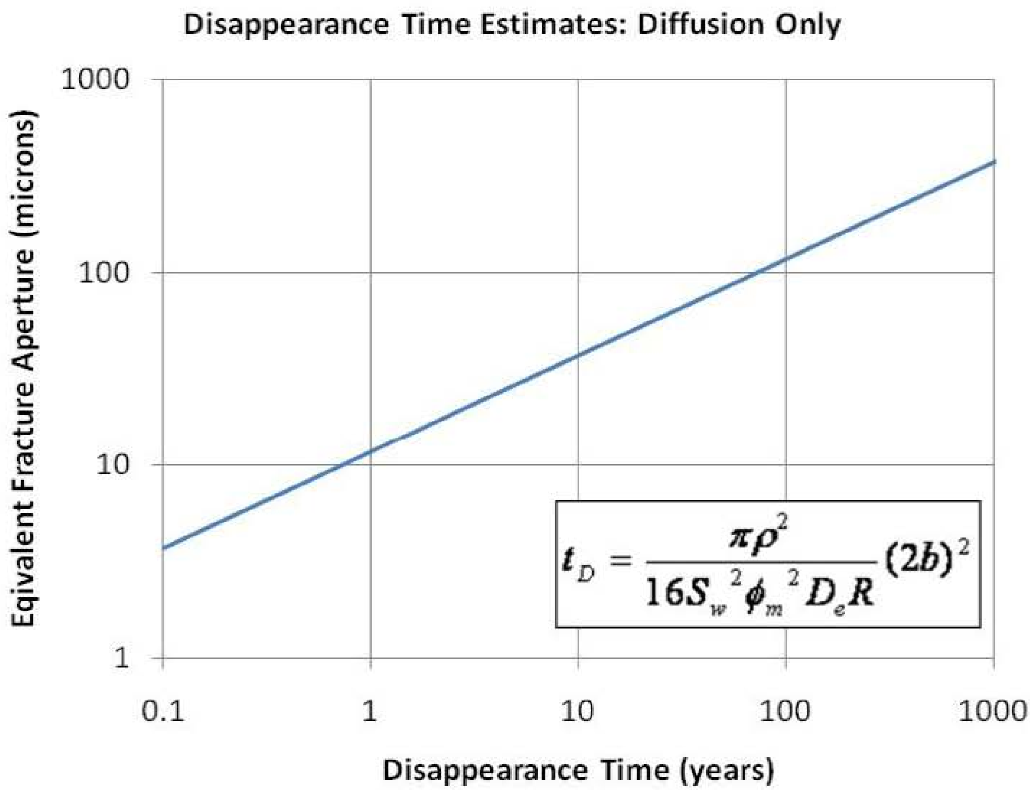
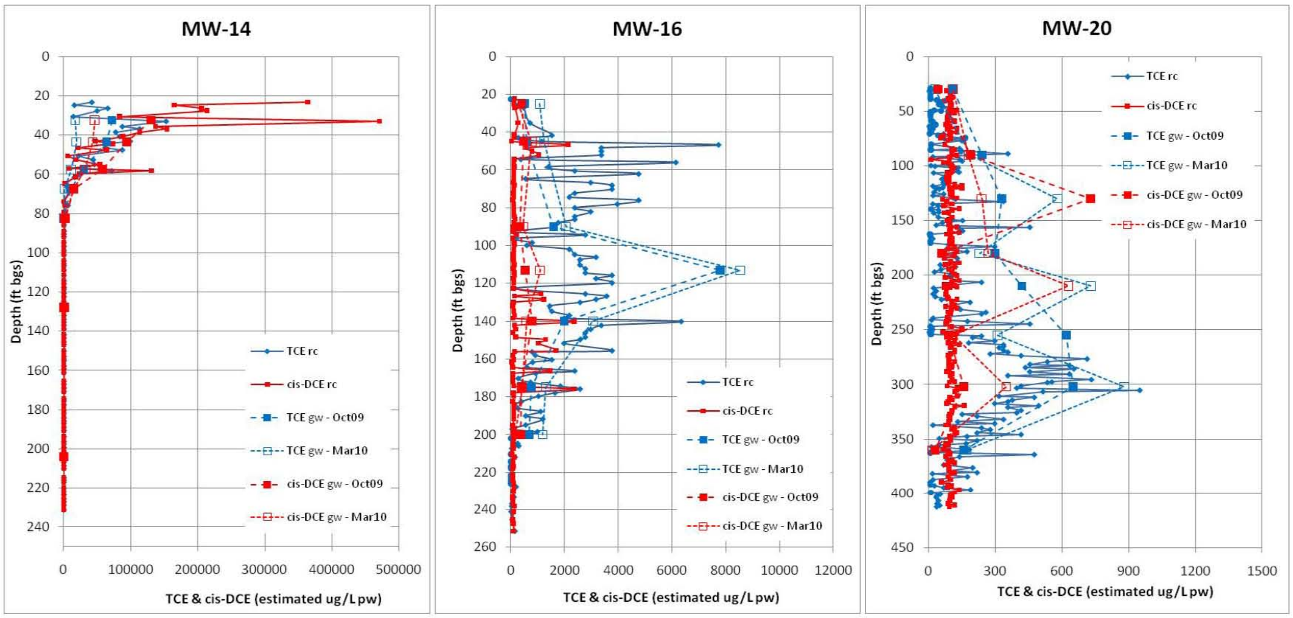


Figure 7. Plot showing equivalent fracture aperture versus DNAPL disappearance time based on diffusion only applying average site matrix parameters (based on methodology in Parker et al., 1994).

(a)



(b)

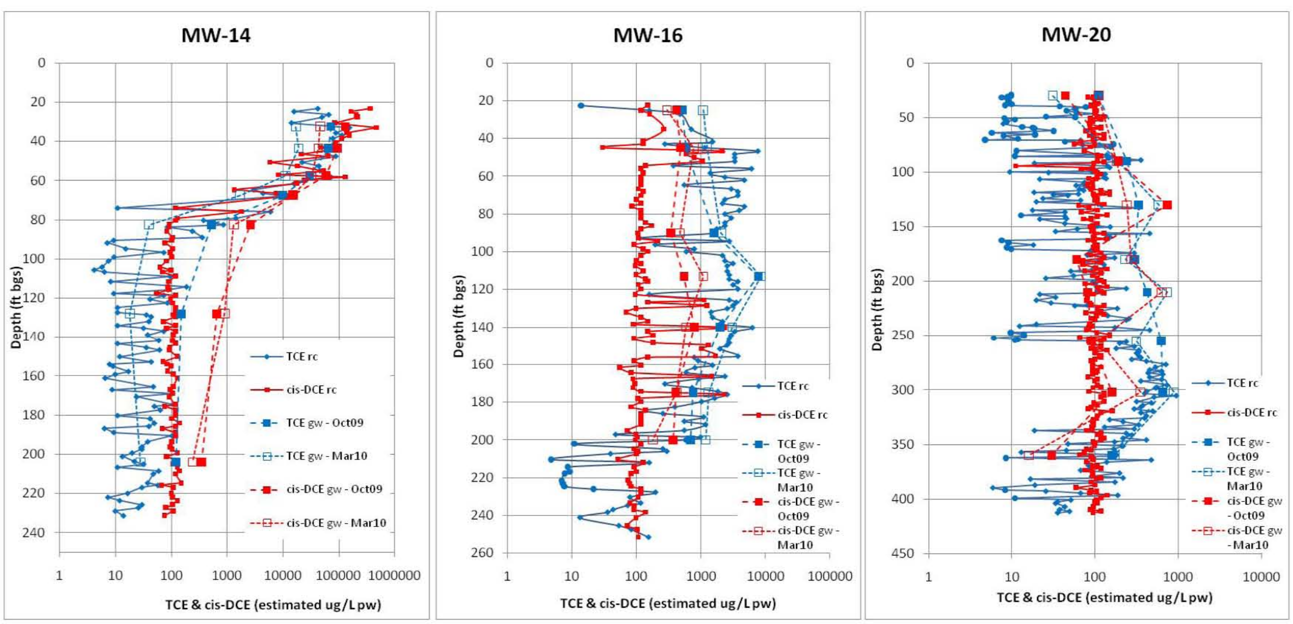


Figure 8. Rock core VOC profiles along the plume flowpath showing estimated TCE and cis-DCE porewater concentrations at MW-14, MW-16 and MW-20 along with groundwater concentrations from two sampling events of the FLUTe multilevel wells installed in these holes, plotted on (a) linear, and (b) logarithmic concentration scales.

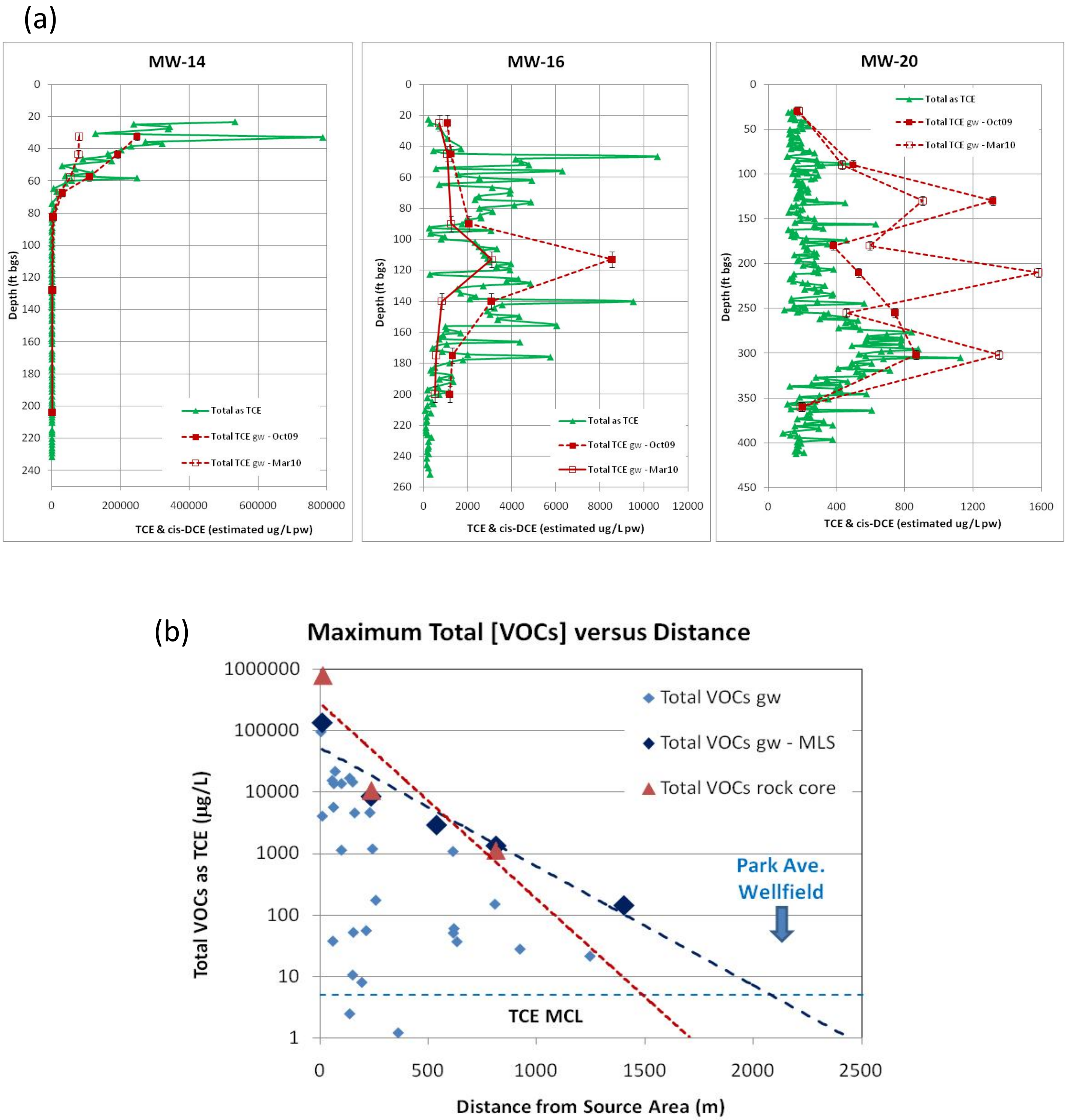


Figure 9. Plots of (a) equivalent TCE concentrations at MW-14, MW-16 and MW-20 from rock core sampling and groundwater sampling of FLUTe multilevel wells installed in these holes, and (b) maximum equivalent TCE versus distance from the site from rock core data and FLUTe multilevel well data along the plume centerline. Data from other monitoring wells and FLUTe multilevel wells are also shown but not used in the interpolations.

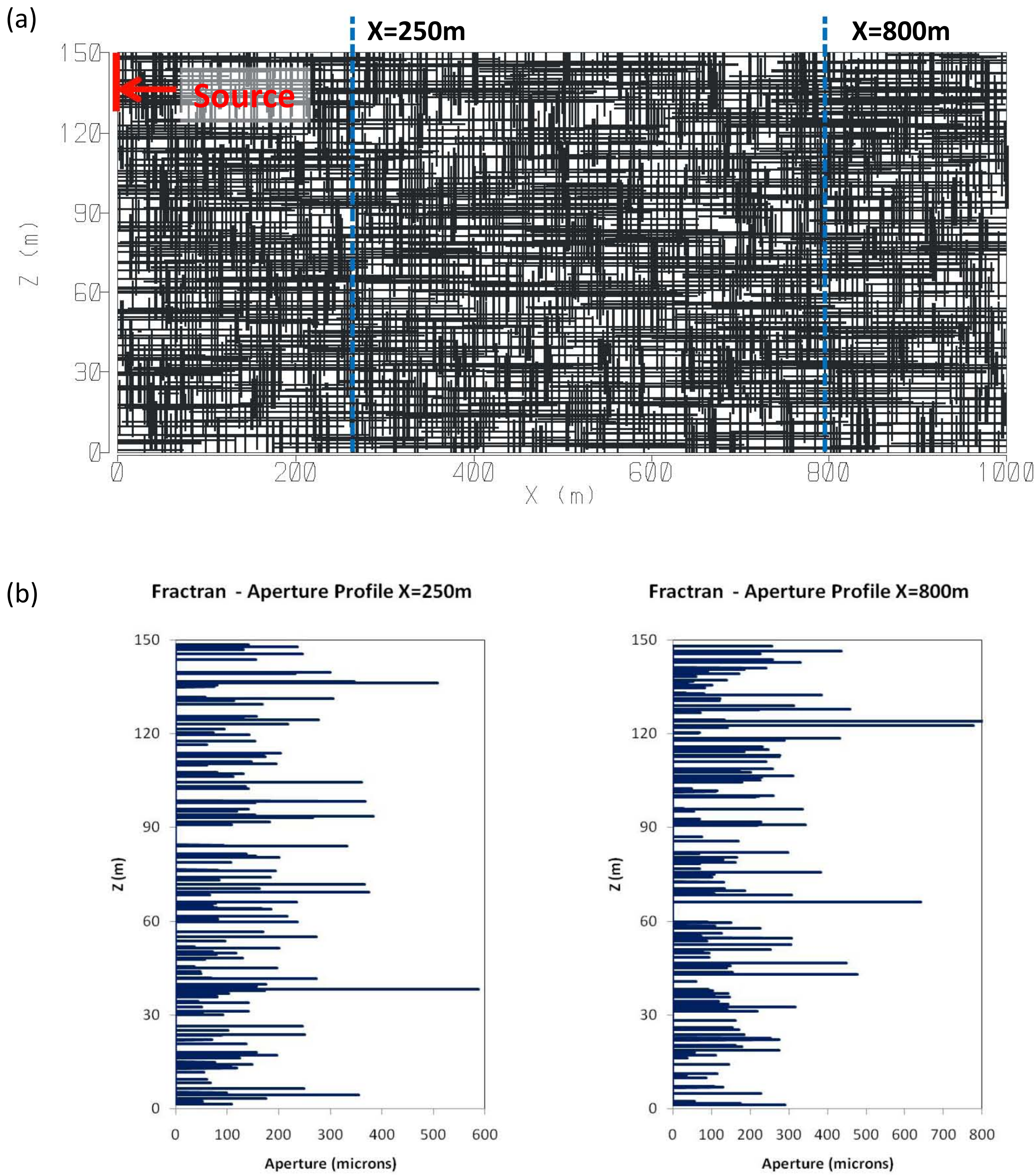


Figure 10. Plots showing (a) FRACTRAN model domain and fracture network, and (b) example profiles showing fracture positions and apertures at X=250 m and 800 m.

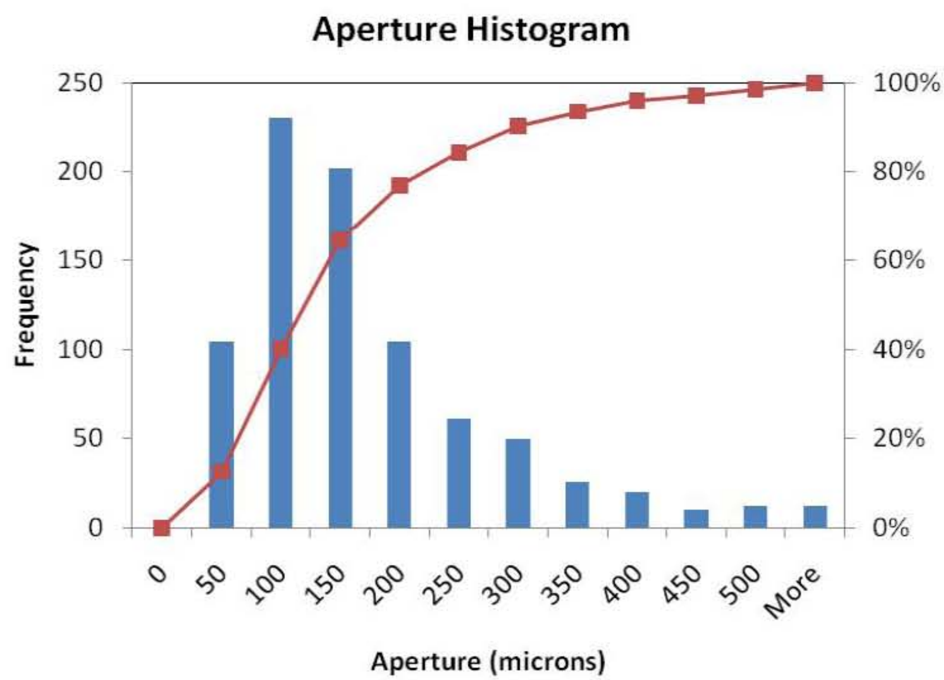


Figure 11. Histogram of horizontal fracture apertures used in the FRACTRAN simulation.

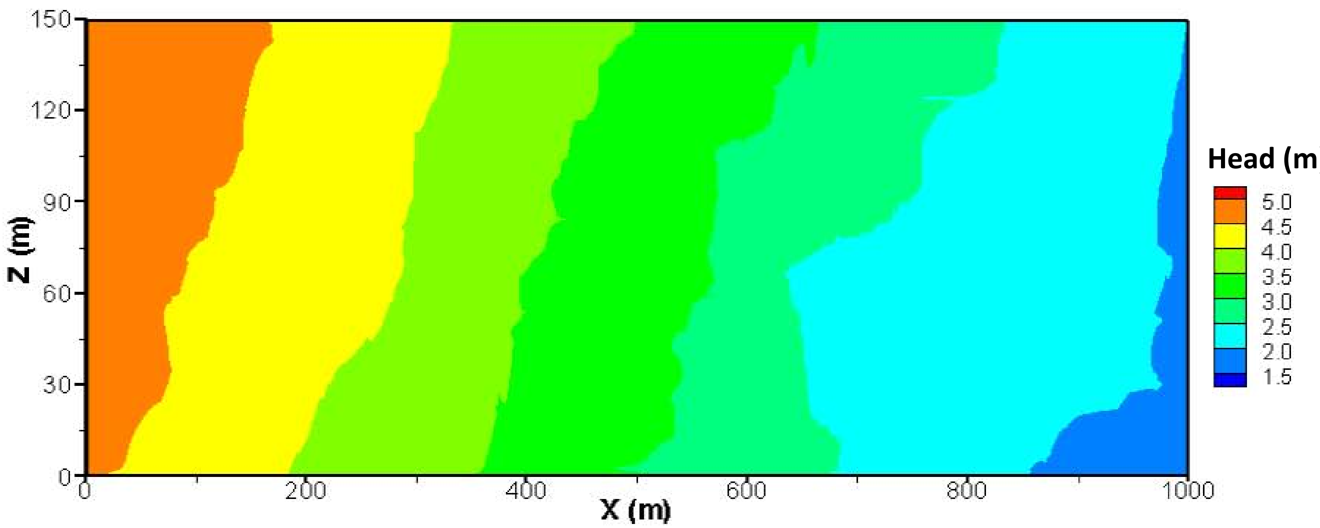


Figure 12. FRACTRAN flow simulation results showing simulated potentiometric surface. Average hydraulic gradients are 0.3% horizontal and 0.3% vertical (downward).

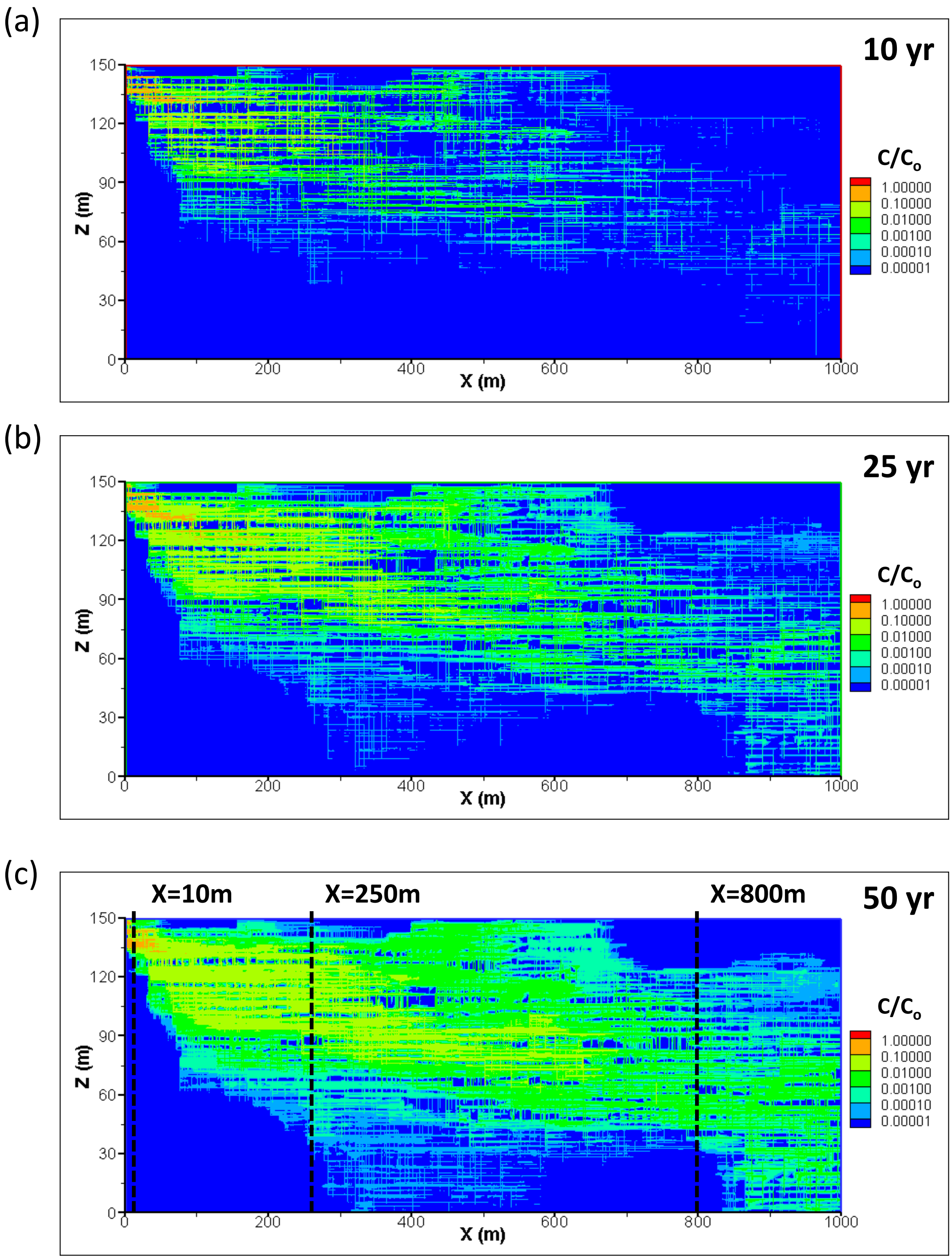
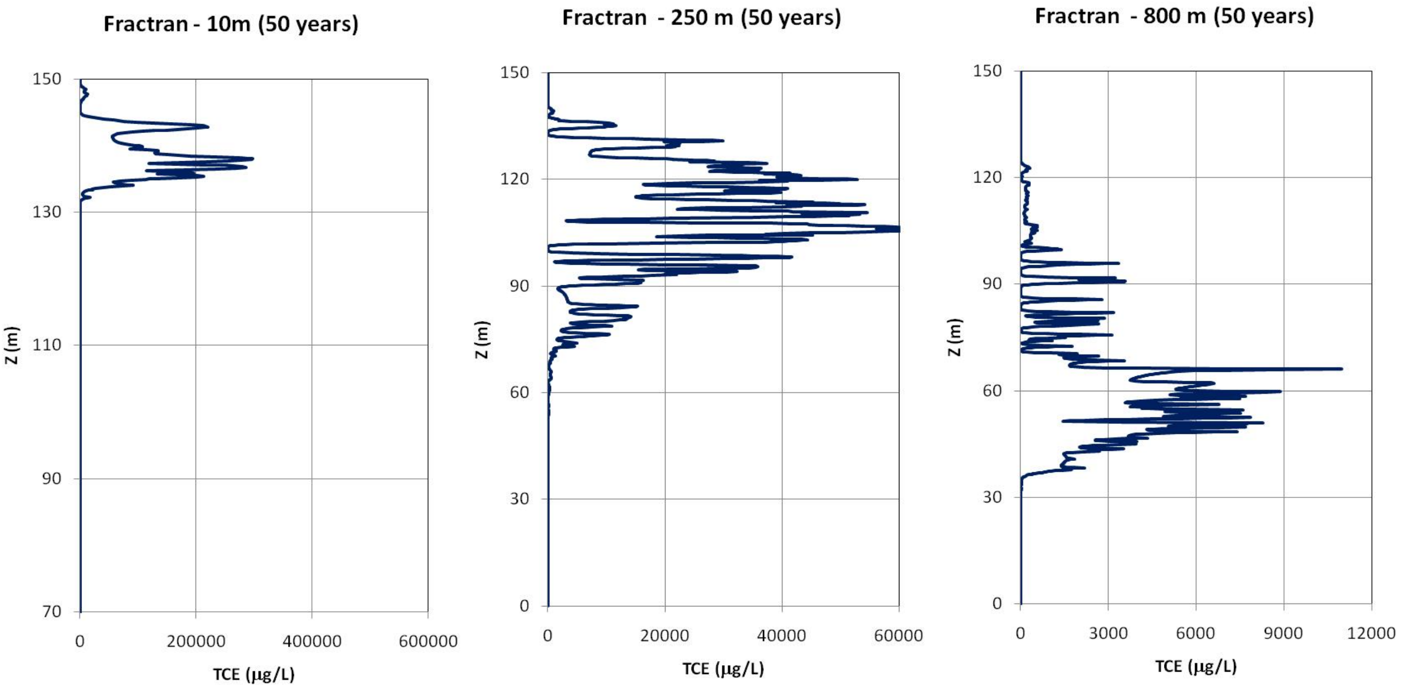


Figure 13. FRACTRAN simulated contaminant plumes at 10, 25 and 50 years plotted as relative concentrations over a 5 order of magnitude range.

(a) FRACTRAN simulated profiles at 50 years



(b) Field rock core profiles

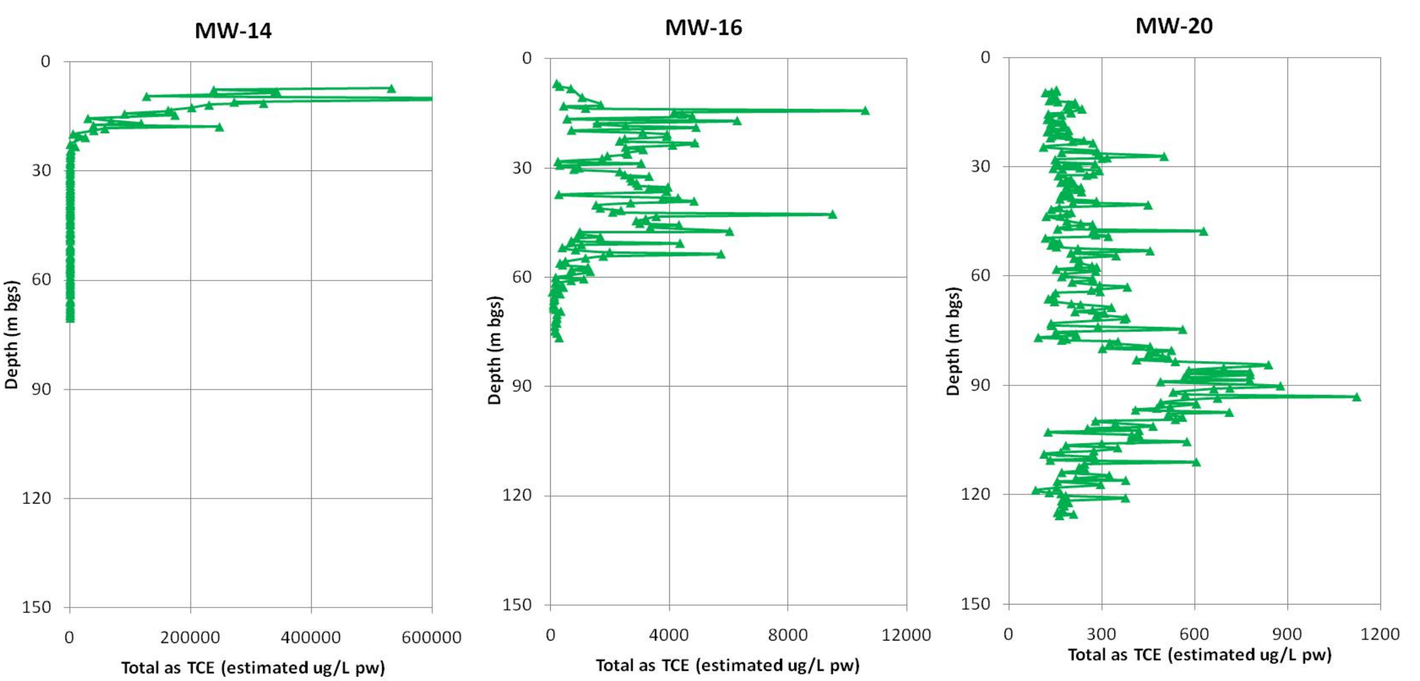
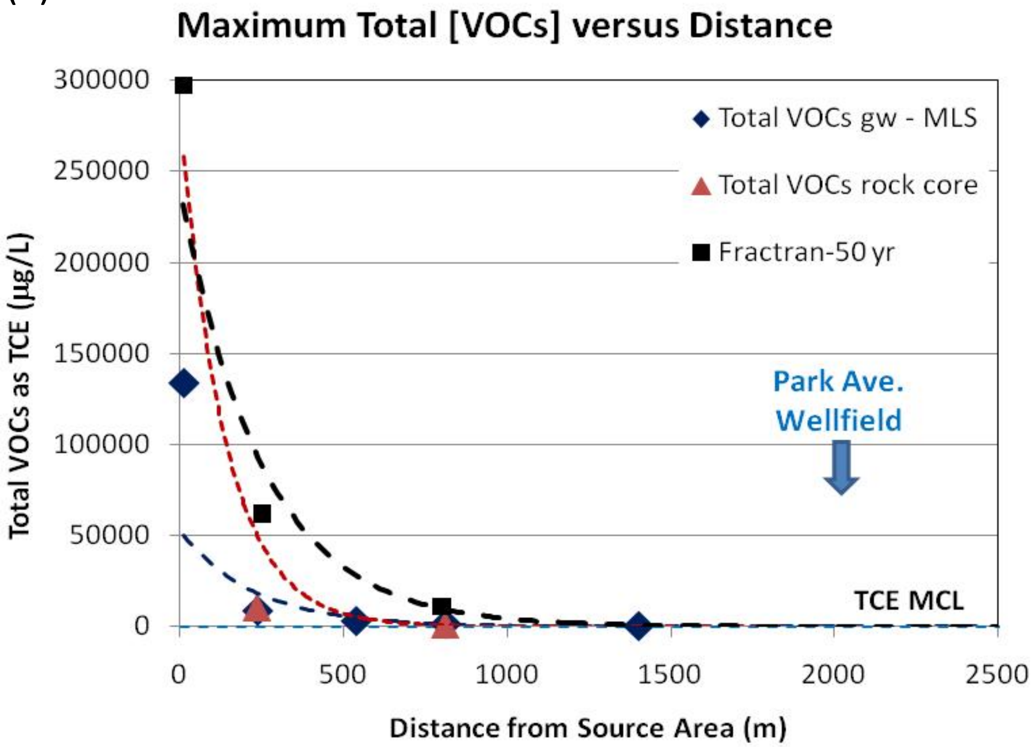


Figure 14. Comparison of (a) FRACTRAN simulated versus (b) field rock core profiles (equivalent porewater TCE concentrations) showing good 'stylistic' comparison. Note the different concentration scales for the MW-16 (5X lower) and MW-20 (10X lower) field profiles compared to FRACTRAN profiles.

(a)



(b)

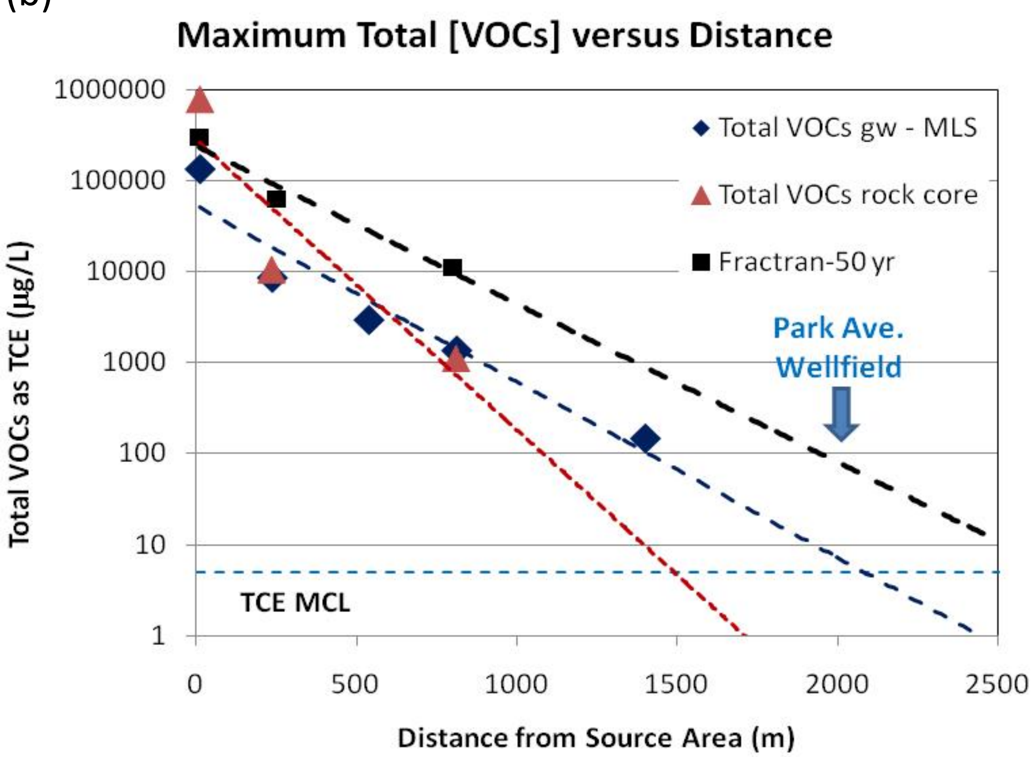


Figure 15. Plots of maximum equivalent TCE with distance from the site on (a) linear and (b) logarithmic concentration scales comparing field data (from rock core VOC sampling and from FLUTe multilevel well sampling along the plume centerline) with the FRACTRAN simulation results.

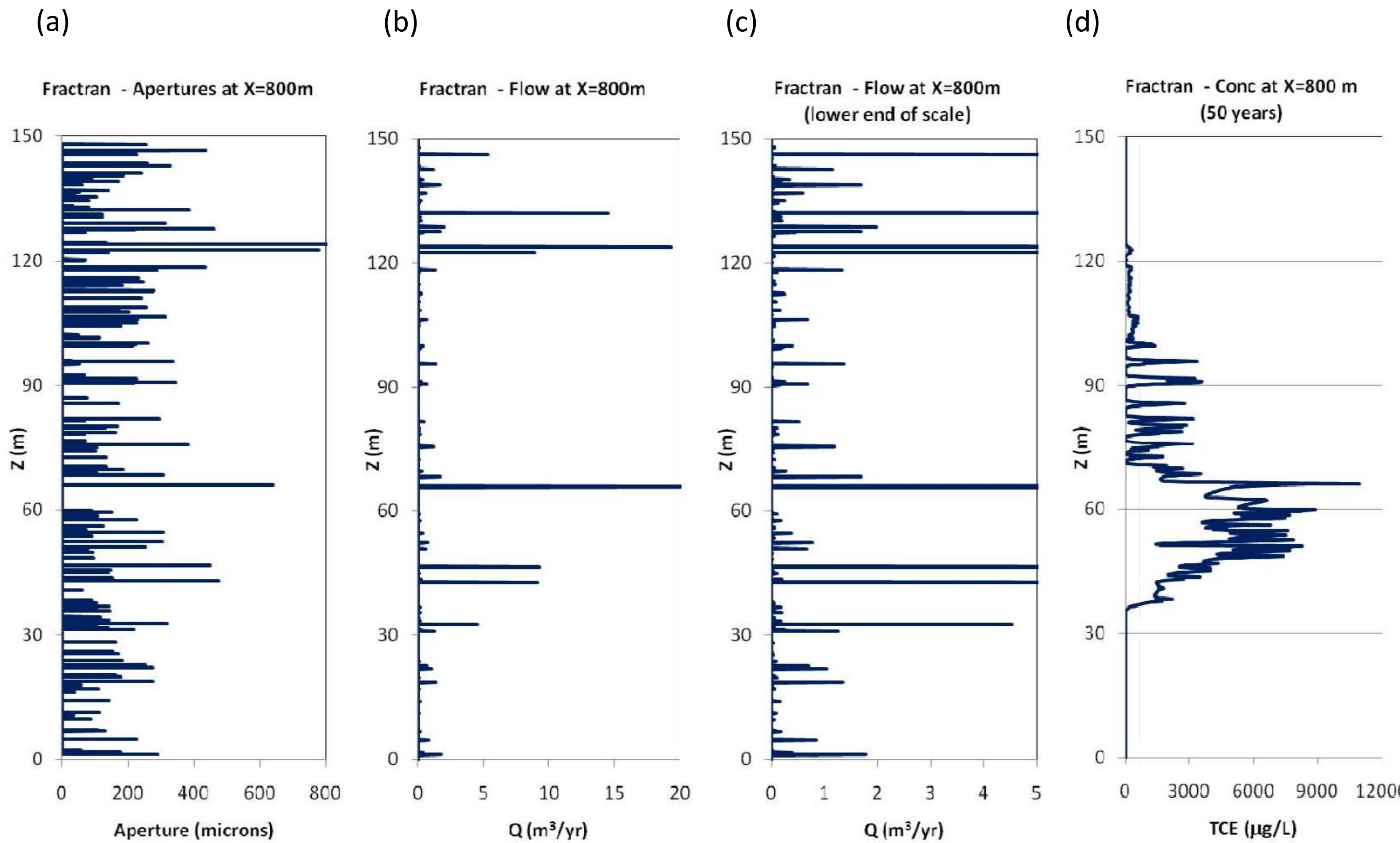


Figure 16. Example of FRACTRAN results at X=800 m showing profiles of (a) fracture positions and apertures, (b) groundwater flow rates, (c) groundwater flow rates on an expanded scale to better show the lower end, and (d) simulated contaminant concentrations at 50 years.

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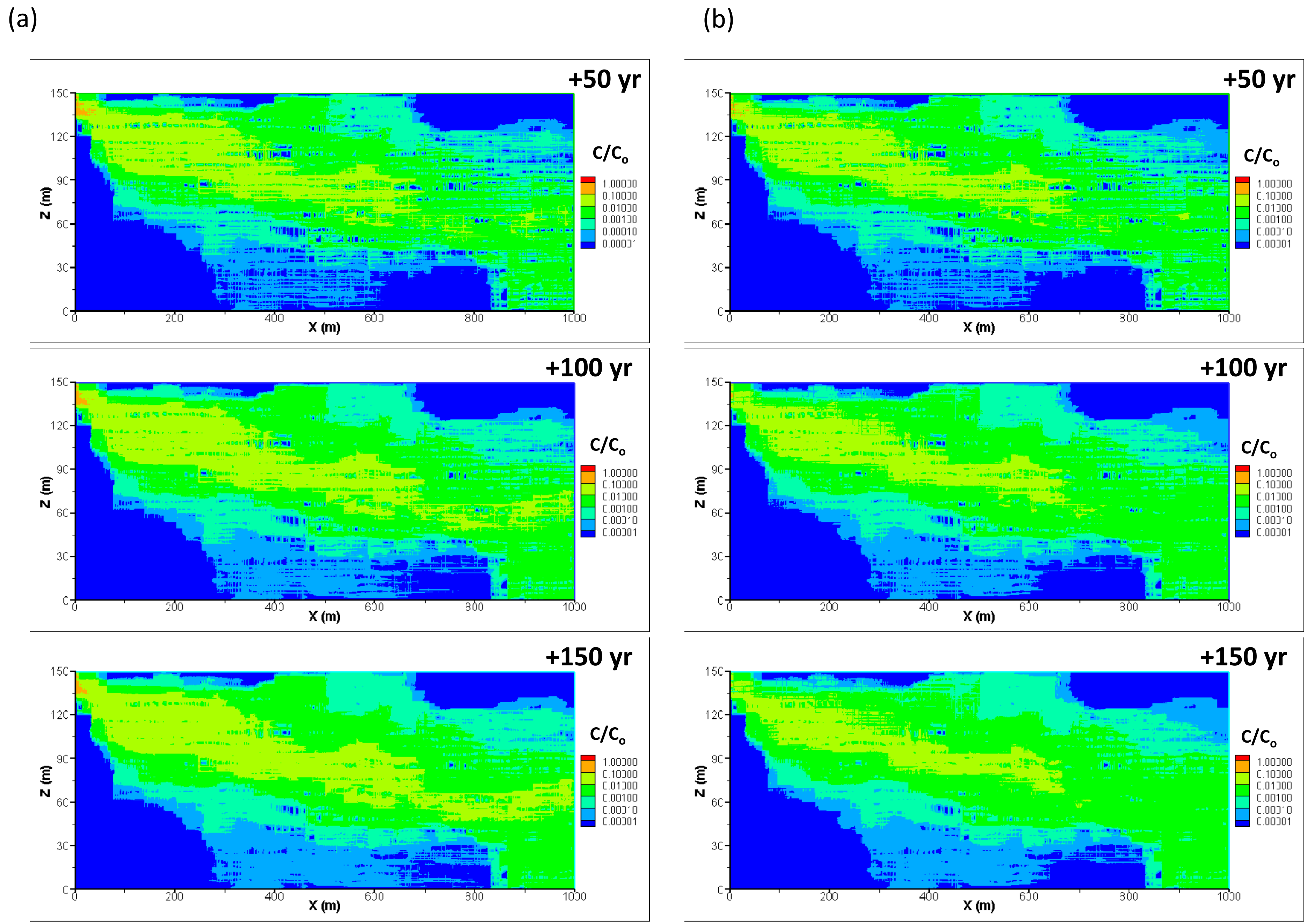


Figure 17. FRACTRAN simulated contaminant plumes at 50, 100 and 150 years from present time for two scenarios of future source inputs: (a) continued input at 10% of solubility, and (b) with complete termination of source input.

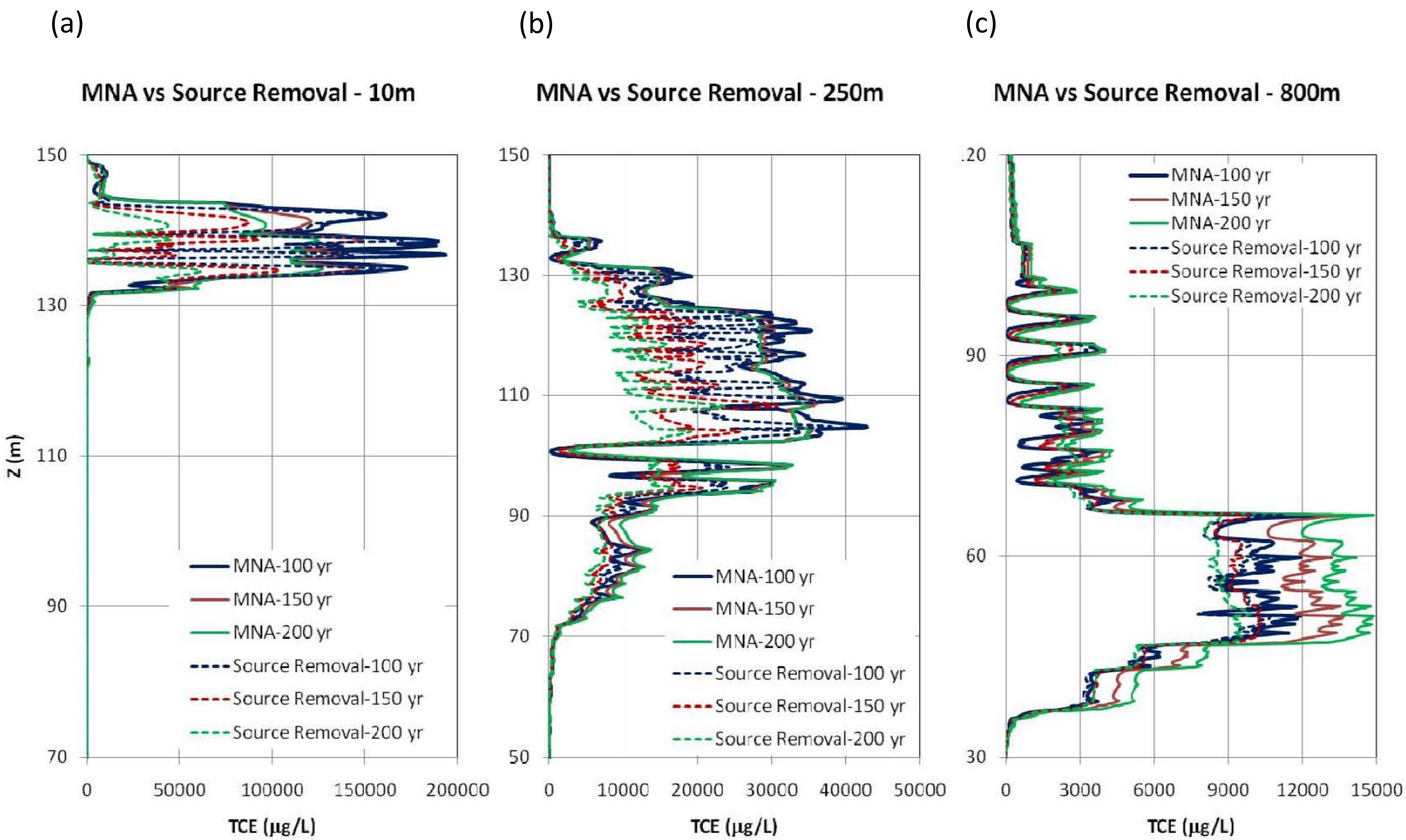


Figure 18. Comparison of FRACTRAN simulated concentration profiles at (a) X=10 m, (b) X=250 m and (c) X=800 m for the two scenarios of future source inputs: (a) continued input at 10% of solubility, and (b) with complete termination of source input.

TABLES

Table 1: Results of physical property analyses on core samples.

Sample ID	Location	Top Depth (ft bgs)	Bottom Depth (ft bgs)	Lithology	Porosity (-)	Water Content (%)	Wet Bulk Density (g/cm^3)	Dry Bulk Density (g/cm^3)	Specific Gravity (-)	Average TOC (%)
CDEMDMW16043.50PHY	MW16	41	44	Mudstone	0.12	2.2	2.57	2.47	2.81	0.0025
CDEMDMW16063.00PHY	MW16	59	64	Mudstone	0.13	4.9	2.52	2.40	2.75	0.0025
CDEMDMW16083.00PHY	MW16	79	84	Mudstone	0.11	3.9	2.56	2.46	2.78	0.0115
CDEMDMW16103.10PHY	MW16	99	104	Mudstone	0.09	3.2	2.56	2.48	2.72	0.0166
CDEMDMW16129.30PHY	MW16	129	134	Mudstone	0.06	2.2	2.62	2.57	2.74	0.0025
CDEMDMW16143.00PHY	MW16	139	144	Mudstone	0.09	3.3	2.63	2.54	2.80	0.0025
CDEMDMW16163.30PHY	MW16	159	164	Mudstone	0.08	2.3	2.61	2.56	2.77	0.0025
CDEMDMW16183.40PHY	MW16	179	184	Mudstone	0.06	1.7	2.66	2.61	2.77	0.0025
CDEMDMW16202.50PHY	MW16	199	204	Mudstone	0.12	4.6	2.59	2.48	2.82	0.0171
CDEMDMW16220.00PHY	MW16	219	224	Mudstone	0.07	2.2	2.67	2.61	2.80	0.2000
CDEMDMW16241.10PHY	MW16	239	244	Mudstone	0.06	2.2	2.62	2.56	2.73	0.0025
CDEMDMW20036.80PHY	MW20	33	38	Mudstone	0.17	7.2	2.44	2.27	2.74	0.0025
CDEMDMW20060.90PHY	MW20	58	63	Mudstone	0.17	6.3	2.51	2.36	2.83	0.0025
CDEMDMW20085.40PHY	MW20	83	88	Mudstone	0.11	3.9	2.61	2.51	2.82	0.0025
CDEMDMW20104.70PHY	MW20	103	108	Mudstone	0.13	5.0	2.60	2.48	2.85	0.0025
CDEMDMW20125.35PHY	MW20	123	128	Mudstone	0.09	3.8	2.62	2.52	2.79	0.0028
CDEMDMW20143.50PHY	MW20	143	148	Mudstone	0.12	4.6	2.60	2.48	2.82	0.0025
CDEMDMW20165.40PHY	MW20	163	168	Mudstone	0.10	3.5	2.64	2.55	2.83	0.0241
CDEMDMW20186.60PHY	MW20	183	188	Mudstone	0.09	3.9	2.59	2.49	2.75	0.0025
CDEMDMW20204.40PHY	MW20	203	208	Mudstone	0.08	3.3	2.63	2.55	2.78	0.0287
CDEMDMW20225.60PHY	MW20	223	228	Mudstone	0.10	3.4	2.61	2.52	2.81	0.0025
CDEMDMW20246.20PHY	MW20	243	248	Mudstone	0.09	2.9	2.63	2.56	2.80	0.0157
CDEMDMW20267.70PHY	MW20	263	268	Mudstone	0.11	3.9	2.55	2.45	2.76	0.0028
CDEMDMW20287.70PHY	MW20	283	288	Mudstone	0.08	2.8	2.63	2.55	2.78	0.0028
CDEMDMW20308.40PHY	MW20	308	313	Mudstone	0.08	2.8	2.64	2.57	2.79	0.0129
CDEMDMW20330.00PHY	MW20	328	333	Mudstone	0.07	2.5	2.64	2.58	2.77	0.0025
CDEMDMW20352.70PHY	MW20	348	353	Mudstone	0.12	3.7	2.58	2.48	2.81	
CDEMDMW20368.70PHY	MW20	368	373	Mudstone	0.10	3.0	2.64	2.56	2.83	0.0199
CDEMDMW20388.40PHY	MW20	388	393	Mudstone	0.08	2.8	2.62	2.55	2.77	0.0122
CDEMDMW20408.40PHY	MW20	408	413	Mudstone	0.08	2.8	2.63	2.56	2.78	0.0025
CDEMDMW14027.20PHY	MW14	24	29	Mudstone	0.16	7.0	2.52	2.36	2.82	0.0028
CDEMDMW14049.30PHY	MW14	49	54	Mudstone	0.13	5.1	2.55	2.42	2.78	0.0025
CDEMDMW14075.90PHY	MW14	73	78	Mudstone	0.09	2.9	2.62	2.55	2.80	0.0332
CDEMDMW14093.40PHY	MW14	93	98	Mudstone	0.09	3.4	2.62	2.54	2.80	0.0247
CDEMDMW14110.00PHY	MW14	108	113	Mudstone	0.08	2.9	2.64	2.56	2.78	0.0227
CDEMDMW14131.00PHY	MW14	128	133	Mudstone	0.07	1.9	2.67	2.62	2.82	0.0025
CDEMDMW14151.10PHY	MW14	148	153	Mudstone	0.11	3.9	2.65	2.55	2.86	0.0146
CDEMDMW14171.80PHY	MW14	168	173	Mudstone	0.08	2.8	2.66	2.58	2.81	0.0148
CDEMDMW14192.00PHY	MW14	188	193	Mudstone	0.08	2.1	2.62	2.56	2.78	0.0177
CDEMDMW14211.40PHY	MW14	208	213	Mudstone	0.09	2.7	2.60	2.54	2.80	0.0164
CDEMDMW14232.50PHY	MW14	213	228	Mudstone	0.12	4.1	2.54	2.44	2.78	0.0153
				Minimum	0.06	1.7	2.44	2.27	2.72	0.0025
				Maximum	0.17	7.2	2.67	2.62	2.86	0.2000
				Average	0.10	3.50	2.60	2.51	2.79	0.0144
				Average*	0.10	3.5	2.60	2.51	2.79	0.0096

* excluding apparent TOC outlier at MW-16 (219-224 ft)

Table 2: Summary of estimated aperture ranges and bulk hydraulic conductivity and fracture porosity from FLUTe liner descent tests

Bedrock Well Number	Borehole Length[1] (feet)	Number of Fractures[2]	Number of Fractures per foot of Borehole	Aperture Data[3] Minimum (microns)	Maximum (microns)	Geometric Mean (microns)	Standard Deviation	Bulk Hydraulic Conductivity (m/sec)	Bulk Fracture Porosity[4] (-)
MW-13	215	241	1.12	4	504	102	59	4.2E-06	2.4E-04
MW-14S	48	18	0.38	39	434	93	85	3.3E-06	1.3E-04
MW-14D	189	212	1.12	15	421	54	40	1.8E-06	2.3E-04
MW-15S	78	140	1.79	6	477	48	60	5.9E-06	3.7E-04
MW-15D	123	234	1.90	3	318	117	36	1.2E-06	3.2E-04
MW-16	194	268	1.38	8	122	52	23	6.5E-07	2.3E-04
MW-17	220	164	0.75	2	1269	35	109	2.2E-05	1.3E-04
MW-18	220	262	1.19	11	470	64	45	3.1E-06	2.8E-04
MW-19	474	224	0.47	9	401	75	50	1.7E-06	1.3E-04
MW-20	351	221	0.63	4	642	169	58	2.7E-06	1.4E-04
MW-21	481	311	0.65	6	509	69	55	2.8E-06	1.7E-04
MW-22	245	211	0.86	15	417	67	43	2.0E-06	2.1E-04
MW-23	420	585	1.39	5	277	50	33	1.4E-06	2.7E-04
FPW	262	267	1.02	9	456	76	55	4.1E-06	2.9E-04
ERT-1	120	75	0.63	11	962	84	164	3.3E-05	2.7E-04
ERT-2	127	35	0.28	11	680	88	141	9.0E-06	1.2E-04
ERT-3	131	63	0.48	37	885	135	139	2.1E-05	2.6E-04
ERT-4	67	71	1.06	12	628	117	117	1.4E-05	5.2E-04
ERT-5	123	83	0.67	6	447	57	80	4.5E-06	1.7E-04
ERT-6	76	33	0.43	34	694	133	148	1.4E-05	2.4E-04
ERT-7	128	123	0.96	11	455	62	60	3.9E-06	2.4E-04
ERT-8	112	61	0.54	17	565	71	135	1.4E-05	
Average			**0.90**	**13**	**547**	**83**	**79**	**7.7E-06**	**2.4E-04**
Min			**0.28**	**2**	**122**	**35**	**23**	**6.5E-07**	**1.2E-04**
Max			**1.90**	**39**	**1269**	**169**	**164**	**3.3E-05**	**5.2E-04**

1 - Length of the borehole tested during the drop liner test
2 - Number of fractures as interpreted by a change in transmissivity during the drop liner test
3 - Apertures estimated using the cubic law assumming one fracture represented by sequential T values in the FLUTe liner test datasets
4 - Bulk fracture porosity estimated by summing all apertures along the borehole and dividing by the borehole test length

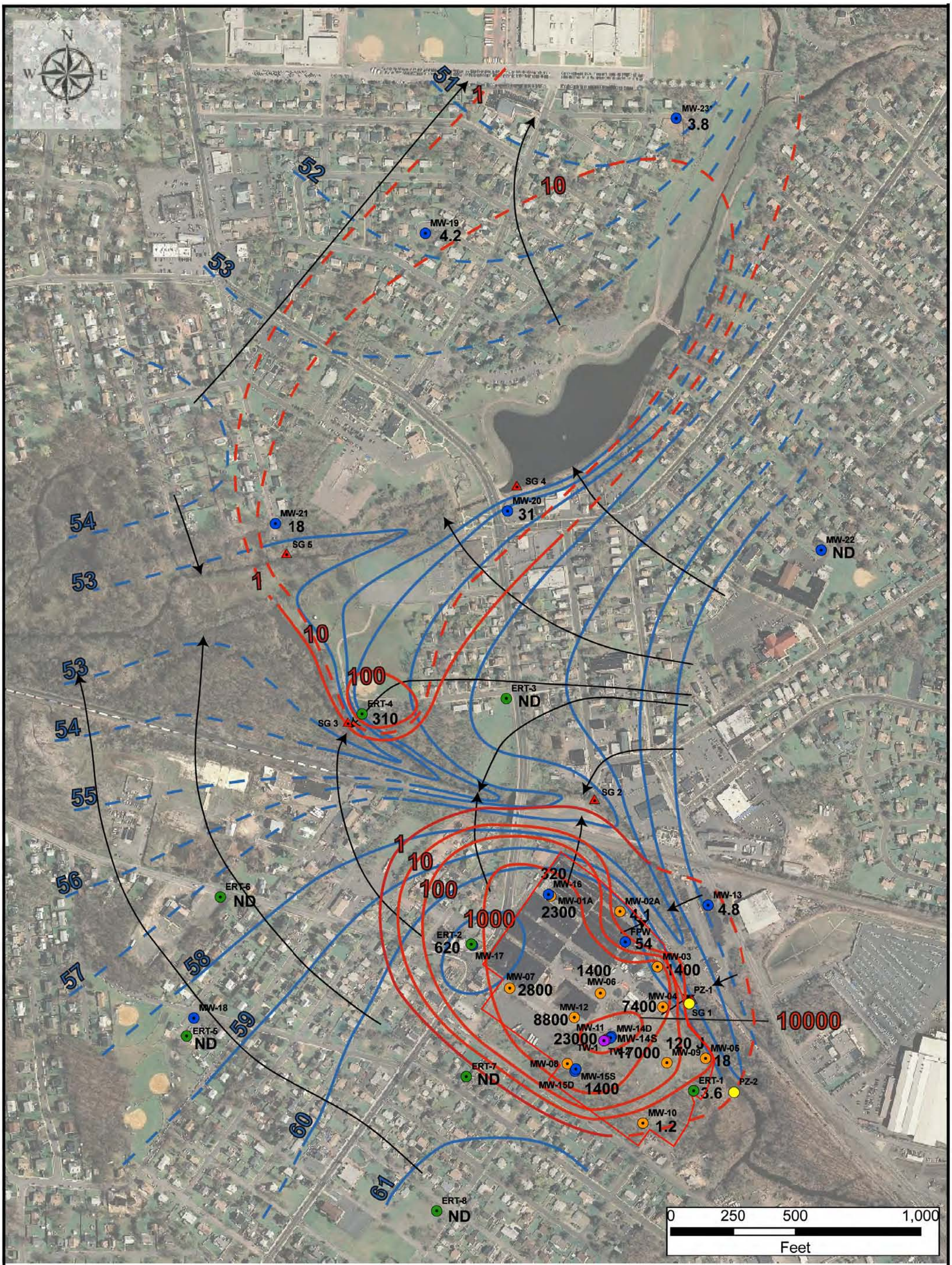
R2-0001557

US Army Corps of Engineers

U. S. Army Corps of Engineers, Kansas City District –

CORNELL-DUBILIER ELECTRONICS SUPERFUND SITE
SOUTH PLAINFIELD, NEW JERSEY
TECHNICAL IMPRACTICABILITY EVALUATION REPORT
OPERABLE UNIT 3: GROUNDWATER

APPENDIX B



Legend

Former CDE Facility	2008 Flute™ Well	Test Well	Piezometer
Shallow Bedrock Monitoring Well	2009 Flute™ Well	Staff Gage	Direction of Groundwater Movement

*Note: MW-23 installed and sampled in December 2010, March 2011

MCL 1 — Line of Equal TCE Concentration (ug/L) (dashed where inferred)

3.6 — Aqueous TCE Concentration (ug/L)

61 — Line of Equal Groundwater Elevation (ft msl) (dashed where inferred)

ARCADIS MALCOLM PIRNIE
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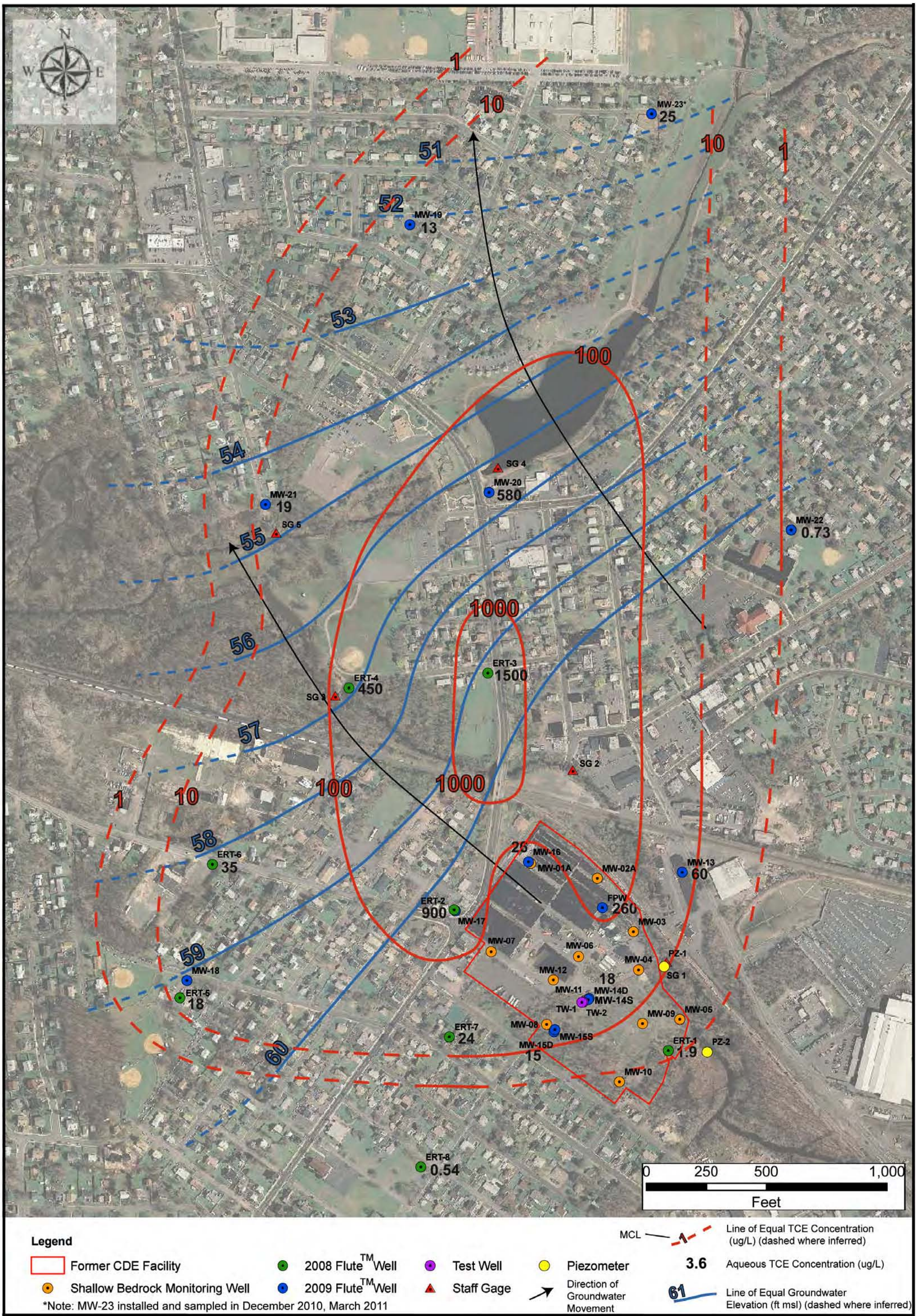
Cornell-Dubilier Electronics
Superfund Site - OU3
South Plainfield, New Jersey

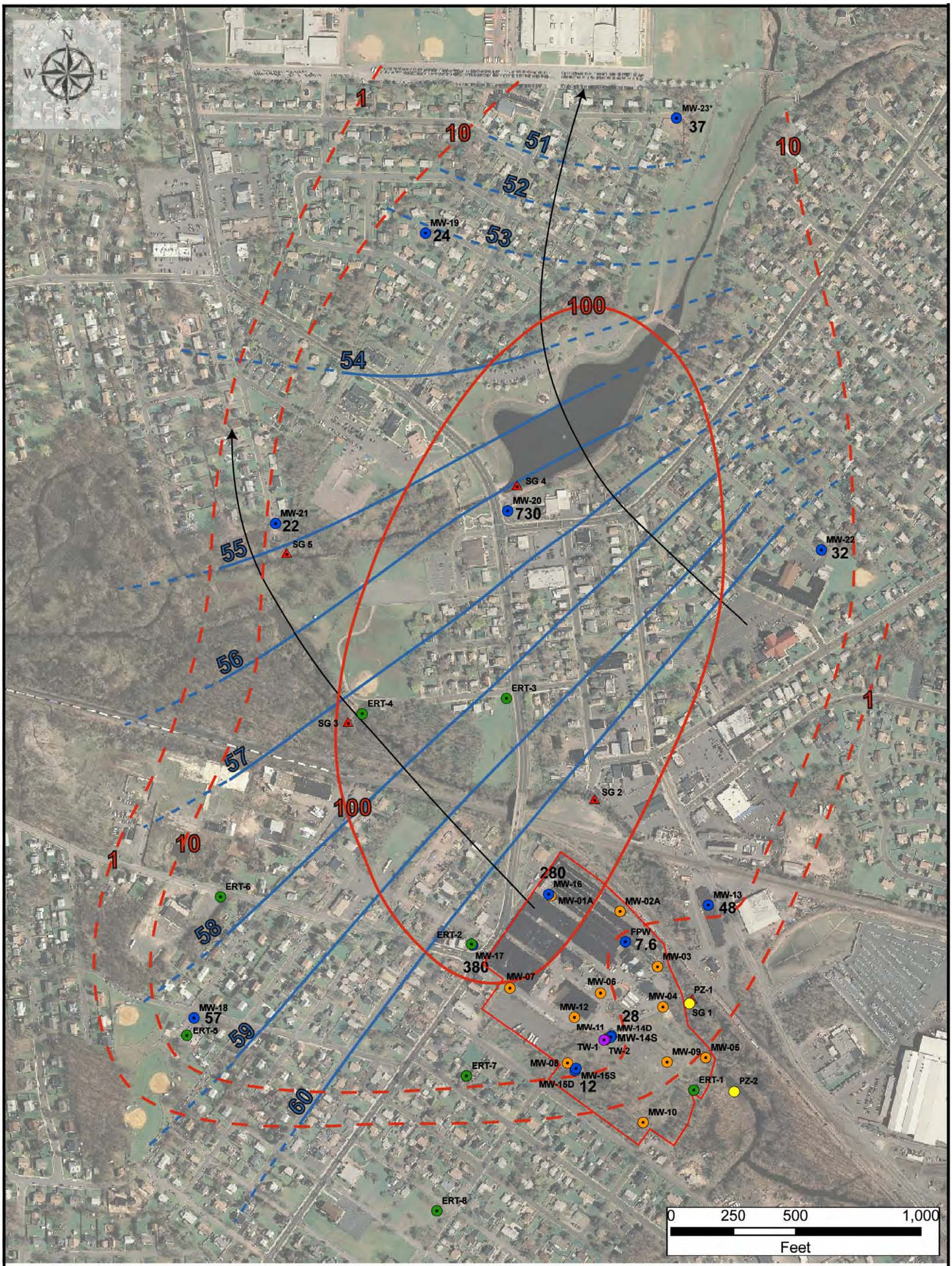
Potentiometric Surface of Shallow (0' - 120' bgs) Water Bearing Zone
July 9, 2010
Aqueous Concentration of TCE
March 2010

Figure 5-11



ARCADIS MALCOLM PIRNIE	Cornell-Dubilier Electronics Superfund Site - OU3 South Plainfield, New Jersey	Potentiometric Surface of Intermediate (120'-160' bgs) Water Bearing Zone July 9, 2010 Aqueous Concentration of TCE March 2010	FIGURE 5-14

R2-0001560



Legend

Former CDE Facility	Shallow Bedrock Monitoring Well
2008 Flute™ Well	2009 Flute™ Well
Test Well	Staff Gage
Piezometer	Direction of Groundwater Movement
MCL — 1	Line of Equal TCE Concentration (ug/L) (dashed where inferred)
3.6	Aqueous TCE Concentration (ug/L)
61	Line of Equal Groundwater Elevation (ft msl) (dashed where inferred)

*Note: MW-23 installed and sampled in December 2010, March 2011

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Cornell-Dubilier Electronics Superfund Site - OU3
South Plainfield, New Jersey

Potentiometric Surface of Deep (200' - 240' bgs) Water Bearing Zone
July 9, 2010
Aqueous Concentration of TCE
March 2010

FIGURE 5-17